

Thesis/
Reports
Rinne, J.N.



VERDE RIVER NATIVE FISHES

THE IMPACTS OF ABIOTIC AND BIOTIC FACTORS

FINAL REPORT FOR HERITAGE PROJECT I96002

JUNE 1996-MAY 1998

JOHN N RINNE AND JEROME A. STEFFERUD

OCTOBER 5, 1998

National FS Library
USDA Forest Service

NOV 21 2011

240 W Prospect Rd
Fort Collins CO 80526

INTRODUCTION

Three objectives for the overall study which encompassed the time period of the contact were addressed for the study:

1) Define:

- a) the short-term (5-year) picture of annual size structure of fish populations and
- b) the relative abundance of each species (native versus nonnative) in the upper Verde River fish community.

2) Characterize and determine the utilization of aquatic macrohabitat types by respective species of fishes.

3) Provide a preliminary analysis of effects of discharge and flow patterns on aquatic macrohabitat dynamics, and fish populations.

Two null hypothesis were proposed to be tested:

- I) Natural hydrographs of the upper Verde River have no effect on sustainability of native fishes.
- II) Introduced species of fish have no effect on sustainability of native fishes.

The first null hypothesis was tested based on data collected through Spring 1998. The upper Verde River hydrograph was analyzed to delineate its effects on aquatic habitat, species abundance, fish population structure, and fish community dynamics. The second null hypothesis was tested based on: 1) relative numbers of native and nonnative species in the total fish community, 2) possible overlap of resource use by native and nonnative fishes, and 3) whether nonnative fishes are competing with and perhaps displacing native fishes from aquatic macrohabitats.

To date, 9 papers (7 published and 2 in press) have resulted from studies on the upper Verde River and its fish community. Most of these papers are appended (see Appendix I) and will be used to both draft and complement this final report. In addition, raw data that has been collected during the study period but has not been analyzed thoroughly or drafted for publication, will supplement existing publications and this report. Figures and tables from publications are enclosed in brackets following the respective paper cited. Non-bracketed figures and tables are original to this report and are presented after the text component of the report.

RATIONALE AND SCOPE FOR STUDY

Two factors appear to interact to affect stability and integrity of native fish populations in southwestern North America: 1) alteration of the natural hydrograph through dams, diversions, or pumping, and 2) introduction of nonnative species of fish. In all but a few streams, these factors have eliminated native fish

species and modified community structure and dynamics, often within a few years after the action. Streams where the native fauna remains predominant are a rarity in the Southwest. The exact mechanism that sustains native species in the presence of nonnative fishes is not understood, however, is thought related, in part, to cycles of drought and flood flows that characterize streams in the southwestern region of the United States. Because of the ever-increasing demand for water for a growing populace, and the generally imperiled status of Arizona's native fish fauna, clarification of this question is important at this time.

The Verde River between Sullivan Lake and Sycamore Creek (Figure 1), retains a native fish fauna that is largely intact; six of the eight native species are present in viable numbers, and the two extirpated species have been reintroduced (Stefferd and Rinne 1995). It is the only reach of major desert river in Arizona where flows are minimally altered by dams or diversions. However, an array of nonnative fish are established in the reach, including red shiner, fathead minnow, common carp, green sunfish, smallmouth and largemouth bass, yellow bullhead, and channel and flathead catfish (Stefferd and Rinne 1995 [Tables 1 - 2]). It is undetermined whether the native species are maintaining their populations in the presence of nonnative species, or whether the nonnative species are unable to become dominant because of flow regimes.

In 1994, the Rocky Mountain Research Station in collaboration with the Tonto and Prescott National Forests initiated research to define the relative roles and influence of physical (hydrology, geomorphology) and biological (nonnative

fish, fish population and community dynamics, fish mobility) factors on the sustainability of native fishes in time (10 years) and through space (7 localities over 37 river miles) in the upper Verde River (Fig. 1). Years one to three of the study were complete upon initiation (July 1996) of Heritage Project I96002. Heritage funding greatly enhanced ongoing Forest Service research studies during years four and five of this long-term, 10-year research effort.

HYDROLOGICAL AND BIOLOGICAL EFFECTS

Monitoring the fish community following floods in 1993 and 1995 (100-year and 10-year recurrence events, respectively) indicated that fish population and community structure and aquatic habitat were affected by these natural perturbations. Total numbers of fish were considerably reduced, but in general, the native species appeared to benefit from the flood, whereas nonnative species were negatively impacted (Stefferdud and Rinne 1995, Rinne and Stefferdud 1996a). For example, based on 1993 data, roundtail chub produced a large year-class that is still evident in the population, and relative abundance of red shiner decreased following the 1995 flood (Fig. 5).

Stream channel morphology and aquatic habitats likewise were affected dramatically by flood events in winter 1993-94. Bed material was resorted, the active channel moved extensively laterally in some reaches, and the stream channel was reconstituted through removal of encroaching riparian vegetation. In

general, floods appear to re-invigorate aquatic habitats of the upper Verde River by facilitating optimal spawning conditions (Rinne and Stefferud 1996a). Reduction of the numbers of nonnative species of fishes, post-flood, also appears to have enhanced the native species of fishes. However, these were short-term observations. The critical need is to examine what the long-term effect of these floods and ensuing hydrological regime may have on fish community structure and native/nonnative interactions.

Long-term (> 5 years) studies of native fish populations in low, desert rivers similar in size to the Verde are non-existent. Several long-term studies in smaller streams are ongoing. Aravaipa Creek in southeastern Arizona has a 25-year record of fish community dynamics. Similarly, there are 10-year records of native fishes and habitat associations for several streams in the upper San Francisco and Gila River drainages in New Mexico. Whereas these existing data sets may be useful for comparing effects of floods and droughts on native species, and perhaps for defining aquatic macrohabitat associations, none of the studies are in stream reaches that have a significant nonnative fish component.

Although precipitation and stream hydrographs are stochastic and unpredictable, floods in the upper Verde River appear to occur on about 7 to 10-year cycles (Stefferud and Rinne 1995). Thus, ensuing years (i.e. post 1995) of low flows and drought conditions become more probable and indeed have occurred (1996-98). Such increased probability of low, base flows provides an excellent opportunity to study native versus nonnative fish interactions during reduced

flooding and low flow conditions. These data can then be compared with measurements taken during periods of higher flows and flooding that probably will occur in the next four to six years. As designed, the Verde River research effort presents a unique opportunity to study the relative influences of abiotic (hydrograph and habitat) and biotic (nonnative species) factors on native fish populations.

STUDY AREA AND METHODOLOGY

General location of study sites were chosen based primarily on reasonable access and linear disposition over the 37-mile reach. Specific location was based on channel morphology and types of habitat present. Seven sites (Figure 1, circles) starting at a point 4.8 miles east of Sullivan Lake and ending at the confluence of Sycamore Creek were sampled during 1994 and 1995. Sites are linearly separated by distances of 2.0 to 10.0 miles. All sites are sampled annually in spring (April-May). Two sites, Burnt Ranch and Sycamore Creek, are sampled again annually in autumn (October).

Exact length of each sample reach is dependent on habitat complexity, but ranges from 1,000 to 2,000 linear feet of river. In general, a riffle-pool sequence was selected, then the study site was expanded to include other significant aquatic macrohabitats present. Macrohabitats initially were defined by subjective

descriptions of channel shape and turbulence, substrate composition, and relative position in the stream channel (Rinne and Stefferud 1996b).

Macrohabitats within a reach were sampled for fishes, progressing in sequence from downstream to upstream. Standard sampling methods for riverine fish, including backpack DC electrofishing units, seines, dipnets, and trammel nets were used relative to macrohabitats sampled. All fish captured within a macrohabitat were identified to species and counted. Total length of all specimens ≥ 6 inches were thus obtained. A subsample of individuals < 6 inches were measured, and weighed to define population demographics. Once measured, all fish except for periodic voucher specimens of nonnatives, were returned alive to the stream.

Each aquatic macrohabitat within a sample reach was numbered in sequence and named (e.g., lateral scour pool, mid-channel pool, backwater pool, glide, run, low and high gradient riffles, and edgewater (Rinne and Stefferud 1996b). Length, width, depth, velocity, and substrate for each macrohabitat were estimated and recorded. Gradients were measured with a laser level, substrates using methodology outlined in Bevenger and King (1995), and velocity with a direct readout digital current meter. For future reference and to enable estimates of habitat dynamics within a study reach, a map of the entire site is sketched annually and location of macrohabitats noted.

Stream gages operated by the U.S. Geological Survey at Paulden and Clarkdale (Figure 1, triangles) continuously record discharge. These data were

used to relate discharge and annual streamflow to the dynamics of the fish community.

Our sampling is designed to expeditiously gather data that will describe the fish community and characterize the aquatic habitats. With these data, we were able to initially delineate relative, qualitative macrohabitats (mean length, width, depth, velocity, and substrate). Refinement of definition of aquatic macrohabitats was considered necessary. Accordingly, increased sampling intensity and rigorous statistical analyses have resulted in greater accuracy in classifying aquatic macrohabitats in the upper Verde River (Sponholtz and Rinne 1997).

RESULTS OF STUDY

Objective 1. Define the short-term (5-year), annual size structure of fish populations and relative abundance of species in the total fish community.

Year Class Structure of Species

Size structure of the two suckers and roundtail chub during 1995-97 are in Figures 2-4. Histograms illustrate the abundance of small-sized individuals of both Sonora and desert sucker and roundtail chub immediately (Spring 1994) after the 1993 flood (Stefferdud and Rinne 1995). Respective size classes or cohorts are evident based on annual spring sampling. Low year class strength for all three of

the large-sized native cypriniforms in successive years (1996 and 1997) are attributed to poor reproductive success in those years, increased influence of non-native species, or both.

In Spring 1994, the modal size of 38 spikedace was 60 mm (mean = 54 mm). Modal size of 72 spikedace in Spring 1995 was again 60 mm (mean = 58 mm). Autumn 1995 sampling reveals that successful reproduction occurred based on modal size of 71 spikedace collected (45 cm; mean size, 43 mm). Of 138 spikedace collected in Spring 1996, modal size again was near 55 mm (mean size = 56 mm). Only six spikedace (mean size 52 mm) were taken in Autumn 1996 and none have been collected since at any of the established sampling sites. Longitudinal sampling of the entire upper river in May 1995 and again in 1997 further document the reduction in the spikedace population in the upper Verde. Only 15 spikedace were collected in the total sampling effort in the upper river in May 1997 compared to almost 140 in May 1995. No spikedace were captured in spring 1998 sampling at the seven established sites or in a survey of the entire upper river. Although the species is noted for these characteristic cycles in abundance, continued and intensified monitoring of this population is recommended.

Fish community structure.

The three large-sized, cypriniform species comprised a major component (60%) of the total fish community and 73% of the native fish fauna in spring 1994 (Stefferud and Rinne 1995 [Table 1], Rinne and Stefferud 1996a, Figure 5). By spring 1995, these three species comprised almost 80% of the native fish fauna before dropping down to 60% of the native fish community in Spring 1996. By spring 1997, these three species comprised 92% of the native fish fauna. Speckled dace were essentially absent and spikédace were absent in samples from established sample reaches. Overall, natives comprised greater than 80% of the total fish community between 1994 and 1996 (Rinne and Stefferud 1996a [Table 1, Fig. 1]). However, by spring 1997 the native component of the total fish community had declined markedly to 19% (Table 1).

A dramatic increase in the number of red shiners in spring 1997 sampling was largely responsible for the inversion of relative numbers of natives to nonnatives from spring 1996 to 1997 (Fig. 5). We predicted that 1998 spring (April) sampling would reveal similar population structure of natives and non-native species if no significant flow events occur before that time. By contrast, if a significant (5 +-year) flow event occurs, we predicted overall fish numbers will be yet be low, but proportions will shift again to favor the native fish component of the total fish community. The former occurred and our predictions were correct.

Spikedace numbers were greatest in spring 1994, dropped in 1995, elevated again in 1996, and disappear from the seven established sampling sites in Spring 1997 and 1998 (Table 1). Sampling optimum habitats in the entire upper river from Burnt Ranch to Sycamore Creek in 1995 resulted in 140 spikedace being taken. Similar sampling was not achieved in spring 1996, however, the same sampling regimen in spring 1997 resulted in only 15 spikedace being collected in the entire 60-km reach of river.

Objective 2 Characterize and determine the utilization of aquatic macrohabitats by respective species of fishes.

Characterization of habitat types

Initially (1994-95), aquatic habitats were visually and subjectively defined for sampling purposes (Stefferud and Rinne 1995). Aquatic macrohabitats in the upper Verde were next defined based on direct physical measurement of gradient, velocity, depth, and substrate composition by Rinne and Stefferud (1996b). Measurements were then made and statistically summarized by the previous, initial subjective classifications. Gradients were first estimated 1995 with the use of a laser level. Gradient estimates successively paralleled subjective, lotic habitat classifications: glides < 0.3%, runs > 0.3 to 5%, low gradient riffles (0.6 to 1%) and high gradient riffles > 1% (Rinne and Stefferud 1996b [Fig. 1]).

Pools displayed a high degree of variability in velocity (0-70 cm/sec) and averaged the same as glide habitats suggesting shortcomings in subjective classifications. Pool classification difficulties are attributable to the many (> a dozen) subjective types of pools (e.g. side channel, mid-channel, scour, etc.) that were grouped in the pool category. Mean velocity did increase as one moved up the lotic scale from run through high gradient riffle habitats (Rinne and Stefferud 1996b [Table 1]). Similarly, substrate type changed from a greater percentage of small material (sand-gravel) in glides and runs in comparison to that in low and high gradient riffles (Rinne and Stefferud 1996b [Fig. 2]).

Increased sampling intensity within study reaches was established in 1996 sampling by placing sample transects every three meters within a habitat type (Sponholtz and Rinne 1997). Previously (1994-95), habitat parameters were measured at only three transects, irrespective of length of habitat type. Normally, four to six transects resulted from the more intensive, detailed, 1996 sampling. As a result, data points for depth and velocity increased three-fold. In addition, gradient was measured in all habitats after initially being estimated at only 14 random habitats in 1995 (Rinne and Stefferud 1996b [Figure 1]). Qualitative habitat descriptions in Spring 1996 were used to delineate habitat type based on selected descriptors (Sponholtz and Rinne 1997 [Table 1]). Pebble counts were performed in all macrohabitats, with 30 "hits" used to estimate substrate type. Rigorous statistical analyses employing ANOVA, and Classification and Regression tree (CART) were performed on data (Sponholtz and Rinne 1997).

ANOVA indicated higher variability in data among transects than among individual points, therefore data were averaged for transects prior to statistics being calculated. Results of analysis suggested depth and velocity were not sensitive enough to detect differences between either of the two riffle types (i.e. high and low gradient) or between run and glide habitats. Gradient was the only variable that most consistently (71% of the time), and clearly, defined aquatic macrohabitat type. Further, when used with depth and velocity, it becomes an even more powerful, descriptive variable providing greater than 80% correct classification of aquatic macrohabitat types.

Habitat utilization by native species

Overall, the native fishes were collected in pool habitats in the upper Verde (Rinne and Stefferud 1996b [Figure 3]). This can be attributed, in part, to the predominance of two of the three large species (Sonora sucker and roundtail chub) which inhabit pools and comprised 30 to 50% of the native fauna in 1994 and 1995, respectively (Rinne and Stefferud 1996b [Table 3]). Further, desert sucker, which comprised 25 to 30% of the native fauna, were captured in pools 20% of the time (Rinne and Stefferud 1996b [Figure 4]). Finally, the great variability in water velocity in pool habitats comprised of multiple (i.e. a dozen) pool categories increased the probability that individual fish would be captured in "pool habitat types." Longfin dace were captured primarily in low gradient riffles and runs [Fig.

5a], spikedace in runs and glides [Fig. 5b], and speckled dace in low and high gradient riffles [Fig. 5c] (Rinne and Stefferud Fig. 5b, see also Neary et al 1996.

Habitat utilization by nonnative species

Nonnative species primarily (29%) used glide habitat. Red shiner were taken in 119 habitats, however, larger numbers (i.e. > 10 individuals) generally were taken in glide habitat. Green sunfish, yellow bullhead, and smallmouth bass were collected in 91 habitats comprised of 45% pools, 22 % glides, 20% runs and 13 % riffles. Common carp was collected in pools over half the time. In addition, Rinne and Neary (1997) demonstrated yellow bullhead and smallmouth bass frequented streambank habitat, because of cover. Further, where these nonnatives were present, native species were absent. These studies are ongoing, but suggest possible competition for habitat and/or predation effects of these two nonnative, predatory species on the smaller native species such as longfin dace and spikedace. Unpublished laboratory data using bullhead and smallmouth bass as predators and red shiners as a surrogate for both the small cyprinid species and young-of-year of the three large cyprinids suggests that predation effect on native fish populations potentially could be substantial.

Objective 3 Provide a preliminary analyses of the effects of discharge and flow patterns on channel morphology, aquatic macrohabitat dynamics, and fish populations.

Flow effects on habitat

Based on annual peak discharges and mean annual discharges over the past three decades, flows in the upper Verde River, as typical of southwestern rivers and streams, were highly variable (Stefferud and Rinne 1995, Rinne and Stefferud 1996a). Neary and Rinne (1997) documented that base flows in the upper Verde have increased over the past two decades. Cycles of flood and drought appear to follow a 7-10 year cycle. Although we have no specific data on channel morphology prior to the 1993 flood, morphology was observed to change dramatically after this 75-100 year event.

By comparison, the 10-year event in 1995 did alter aquatic habitat as evidenced in changes in the relative abundance of habitat types (Table 2). Based on aquatic macrohabitat type, most of the major sample reaches do change in relative aquatic macrohabitat from year-to-year (Table 2). Lack of significant flood events following 1995 sampling, or ensuing low, drought flows have had significant effects on macrohabitats (raw data). Stream channels have narrowed and aquatic macrophytes have encroached from stream margins, reducing mean

stream width. Depths of aquatic habitats also have been reduced because of a lack of bankfull, or greater, scouring flows.

Flow effects on fishes

The fish community in the Verde River is very responsive to flow regime (Stefferdud and Rinne 1995, Rinne and Stefferud 1996a). In general, very large, peak flood events such as the 1993, 100-year event, drastically reduce the total fish community. Our sampling did not commence until 1994, but sampling by personnel from the Arizona Game and Fish Department in Spring 1993, immediately following massive flooding that winter, suggests all species were drastically reduced in abundance. Spring 1994 compared to spring 1995 data indicate that the fish community expanded by the former sample date and then dropped after the less dramatic flood events of spring 1995 (Rinne and Stefferud 1996a [Table 2]). Sampling in 1996 documents native species rebounding, along with red shiner (Fig. 5). Continued low flows and lack of significant flooding in 1996, and prior to 1997 Spring sampling, resulted in a dramatic increase in red shiner, smallmouth bass and yellow bullhead (Table 1, Rinne and Stefferud 1996a [Table 2], Fig. 5).

In summary, the total fish community in the upper Verde definitely responds to the hydrograph. That is, very large floods cause a marked reduction in both native and nonnative species. Native species rebound very quickly apparently as a

result of optimal spawning and recruitment (Figs. 2-4) and reduced numbers of nonnatives. Nonnatives then increase relative to natives as lack of flooding and base or drought flows are sustained.

CONCLUSIONS

Null hypothesis I obviously has to be rejected based on the response of fishes in the upper Verde River relative to flood and drought flows. Null hypothesis II also has to be rejected based on both relative community composition and the utilization of streambanks by nonnative species and their probable predatory effects on native fishes.

The relative influences of abiotic versus biotic (i.e. hydrological regime vs nonnative fishes) cannot be unequivocally stated at this time. We have suggested (Rinne and Stefferud 1996a) that the abiotic factors (flows and habitat) are of primary importance and the controlling factors in sustaining native fishes in the upper Verde. Introduced fishes (biotic) and their interactions with native species are of perhaps secondary significance in legislating numbers, distribution and sustainability of native fishes in the upper Verde River. Probably the best interpretation is that the interactions of the two are actually the real process. That is, flood flows reduce the overall fish community in numbers and distribution, however, the native species appear to recover and predominate the fish community immediately post-flood. The cessation of flood events then results in nonnative

species becoming, steadily and increasingly, more abundant (Rinne and Stefferud 1996aa, Table 1). Once nonnatives predominated the fish community, the biotic factor may then become the variable that controls or delimits fish community structure.

The size of the flow event necessary to effect a significant change in fish community structure cannot be defined based on data collected to this point in time. Obviously, 100-year events are more than adequate to reduce or remove nonnative species. Ten-year events also have an effect based on our data from the upper Verde. The threshold below which flow events have no significant impact on the nonnative proportion of the fish community and their predominance becomes the controlling factor on native fish populations (through more successful reproduction, recruitment, predation, and competition for food and space) is not offerable at this time. The stochasticity of flows over the next five years may aid in refinement of this definition.

DISCLAIMER

The findings, opinions, and recommendations in this report are those of the investigators who have received partial funding from the Arizona Game and Fish Department Heritage Fund. The findings, opinions, and recommendations do not necessarily reflect those of the Arizona Game and Fish Commission or Department,

or necessarily represent official Department policy or management practice. For further information, please contact the Arizona Game and Fish Department.

LITERATURE CITED

Bevenger, G. S. and R. M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. USDA Forest Service Research Paper RM-319:1-17.

Neary, A. P., J. N. Rinne, and D. G. Neary. 1996. Physical habitat use by spikedace in the upper Verde River, Arizona, pp 23-28. In, Hydrology and Water Resources in Arizona and the Southwest. Vol. 26. Proc. of the 1996 meetings of the Ariz. Section, Amer. Water Resour. Assoc. and the Hydrology Section, Arizona-Nevada Acad. of Sciences, April 20, 1996, Tucson, Arizona.

Neary, D. G. and J. N. Rinne. 1997. Baseflow trends in the upper Verde River relative to fish habitat requirements, pp 57-63. In, Hydrology and Water Resources in Arizona and the Southwest. Vol. 27. Proc. of the 1997 meetings of the Ariz. Section, Amer. Water Resour. Assoc. and the Hydrology Section, Arizona-Nevada Acad. of Sciences, April 19, Las Vegas, Nevada.

- Rinne, J. N. and J. A. Stefferud. 1996a. Factors contributing to collapse yet maintenance of a native fish community in the desert Southwest (USA), pp 157-162. In, D. A Hancock, D. C. Smith, A. Grant, and J. P. Beaumer (eds). Developing and Sustaining World Fisheries Resources: The state of science and management. Second World Fish Congress, Brisbane, Australia. July 28-Aug 2, 1996.
- Rinne, J. N. and J. A. Stefferud. 1996b. Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona, pp 13-22. In, Hydrology and Water Resources in Arizona and the Southwest. Vol. 26. Proc. of the 1996 meetings of the Ariz. Section, Amer. Water Resour. Assoc. and the Hydrology Section, Arizona-Nevada Acad. of Sciences, April 20, 1996, Tucson, Arizona.
- Rinne, J. N., and D. G. Neary. 1997. Stream channel and fish relationships: Preliminary observations, Verde River, Arizona, Pp 475-482. In, Proc. AWRA Sympos, Water Resources Education, training, and practice: Opportunities for the next Century.
- Rinne, J. N. J. A. Stefferud, A. Clark, and P. Sponholtz. In press (1998). Fish population structure in a large desert river relative to anthropogenic influence: Verde River, Arizona, USA., pp - . In, International Symposium and Workshop on Management and Ecology of River Fisheries. Mar. 30-April 3, 1998. Hull, England.

Stefferd J. A. and J. N. Rinne. 1995. Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes, pp. 26-32. In, Hydrology and Water Resources in Arizona and the Southwest. Vols. 22-25. Proc. of the 1995 meetings of the Ariz. Section, Amer. Water Resour. Assoc. and the Hydrology Section, Arizona-Nevada Acad. of Sciences, April 22, 1995, Flagstaff, Arizona. 114 pp.

Sponholtz P. and J. N. Rinne. 1997. Refinement of aquatic macrohabitat definition in the upper Verde River, Arizona, pp 17-24. In, Hydrology and Water Resources in Arizona and the Southwest. Vol. 27. Proc. of the 1997 meetings of the Ariz. Section, Amer. Water Resour. Assoc. and the Hydrology Section, Arizona-Nevada Acad. of Sciences, April 19, Las Vegas, Nevada.

Sponholtz, P. D. Redondo, B.P. Deason, L. Sychowski, and J. N. Rinne. In press (1997). The influence of stock tanks on native fishes: Verde River, Prescott National Forest, pp - . In, Proc. Sympos on Environmental, Economic, and Legal Issues Related to Rangeland Water Developments. Nov 13-15, ASU, Tempe, Arizona.

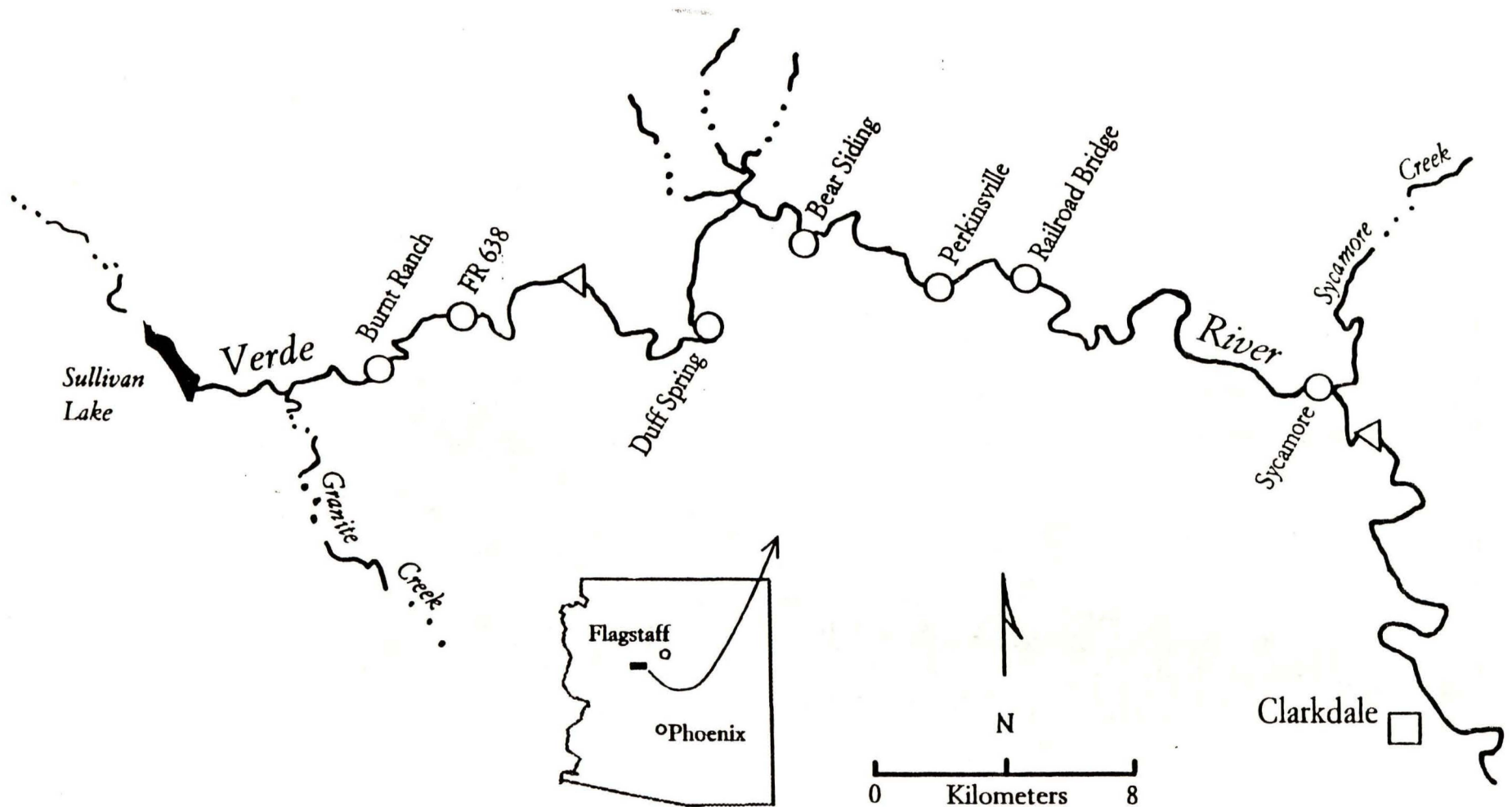


Figure 1. Map of upper Verde River study area indicating the seven established sample sites (circles), and USGS Guages (triangles).

Figure 2. Length frequency of sonora suckers in the upper Verde River during spring 1994 to 1997.

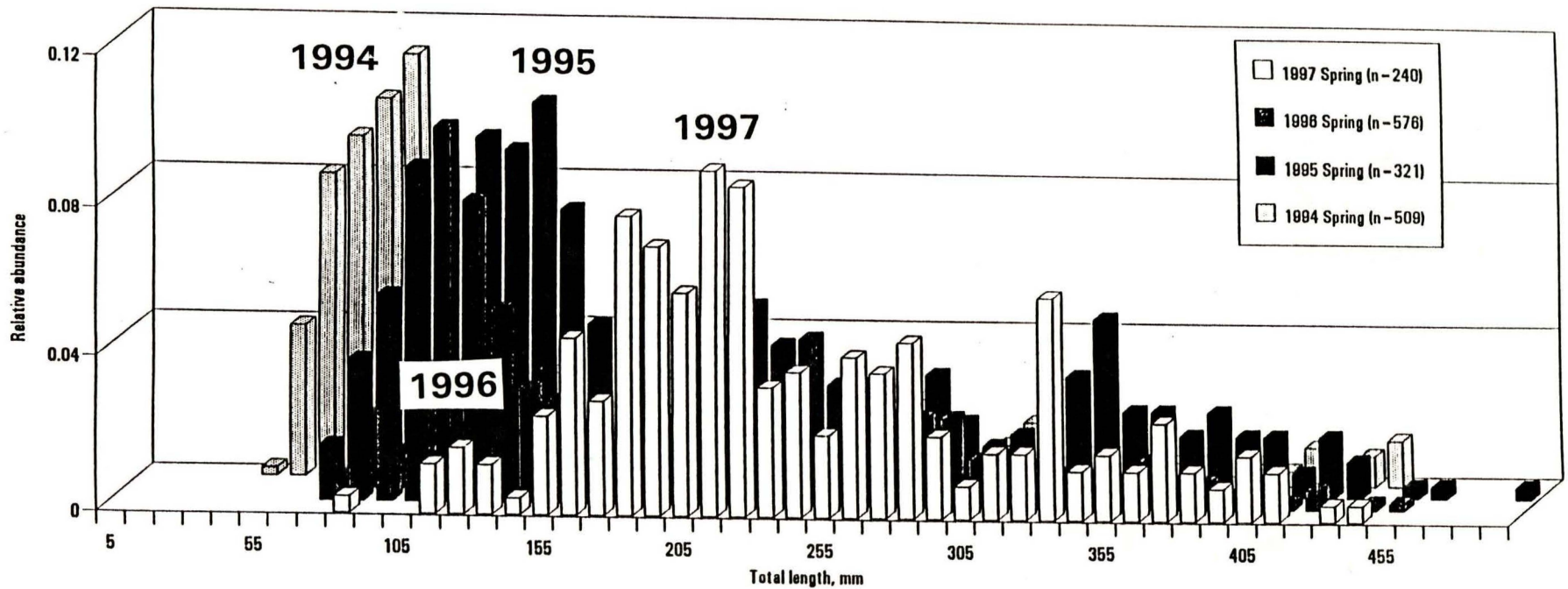


Figure 3. Length frequency of desert suckers in the upper Verde River during spring 1994-1997.

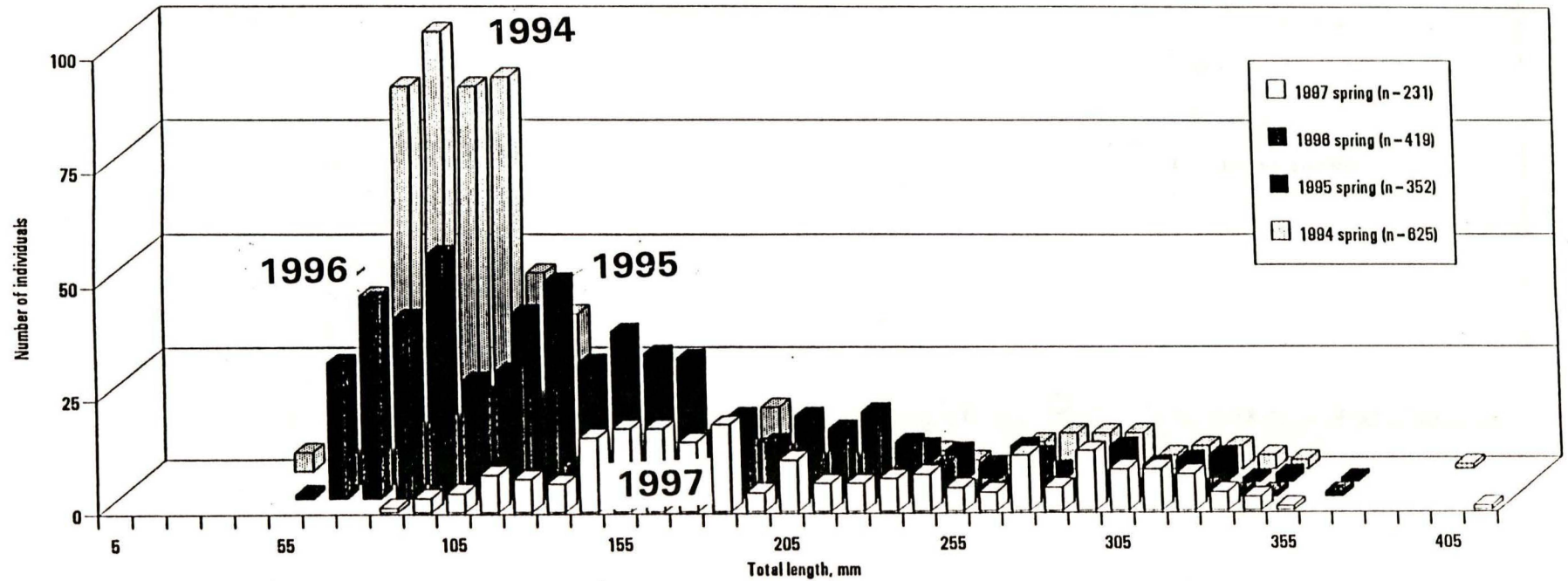


Figure 4. Length frequency of roundtail chubs in the upper Verde River during spring, 1994 to 1997.

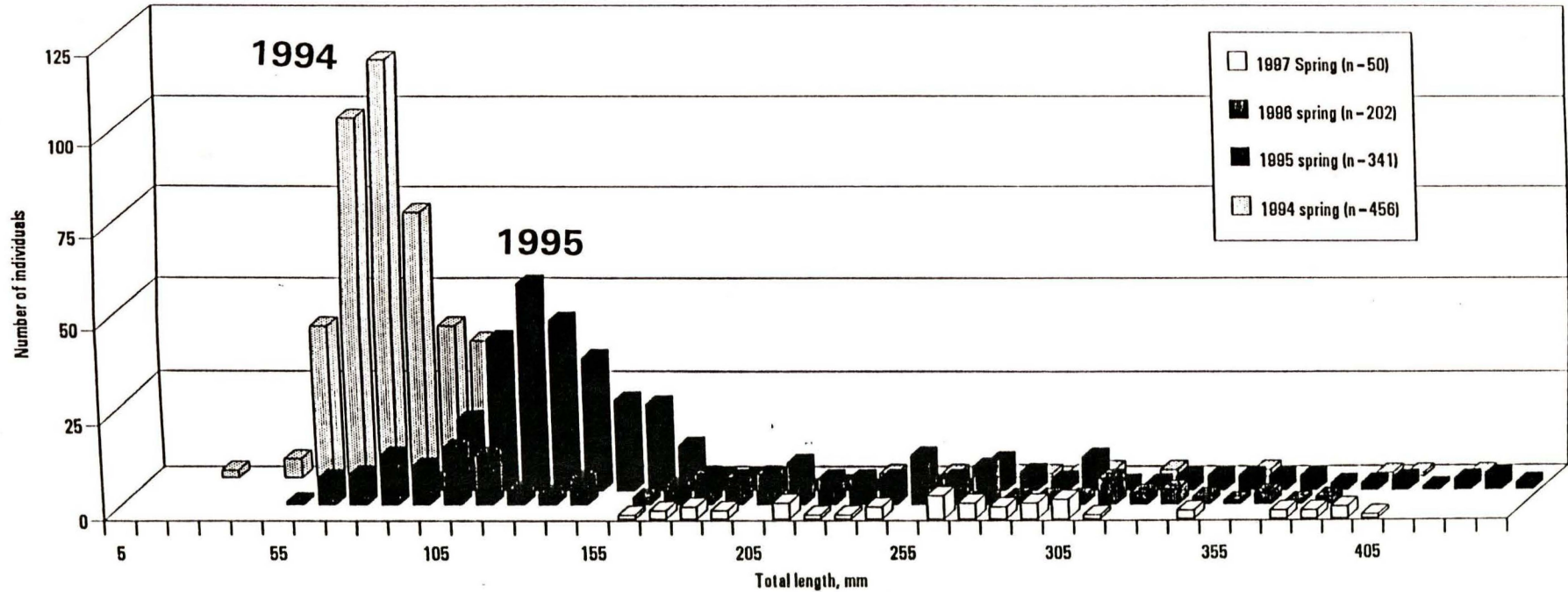
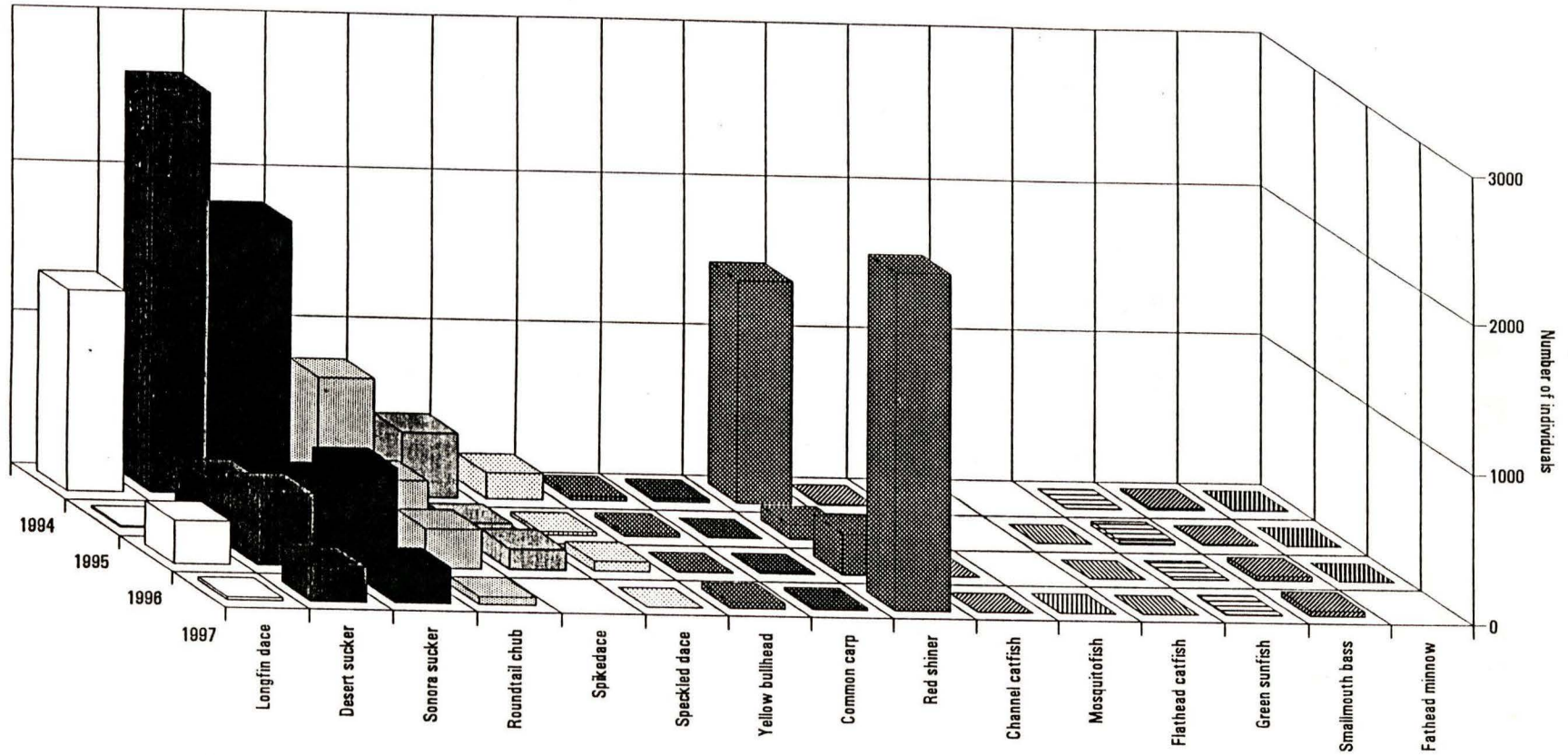


Figure 5. Abundance of fishes at seven regular sampling sites, upper Verde River, 1994-1997.



sampling sites in the upper Verde River during spring, 1994-1997.

A Total number of fish collected at seven sites in upper Verde River, April-May 1994.

Total number of fish collected at seven sites in upper Verde River, April-May 1994.																
	Species															Total
	Longfin dace	Desert sucker	Sonora sucker	Roundtail chub	Spikedace	Speckled dace	Common carp	Red shiner	Fathead minnow	Yellow bullhead	Green sunfish	Smallmouth bass	Channel catfish	Flathead catfish		
Burnt Ranch	1072	339	278	15	257		1	39		2		2			2005	
FR638	227	230	154	18	2			81		3					895	
Duff Spring		192	329	28	1		3	32		1		4			590	
Bear Siding		382	357	249	1	3	4	227		1		3	1		1228	
Perkinsville	19	885	442	244	37	81	11	1109	7	12	4	2	4		2837	
Railroad Bridge		237	27	57	38	88		2		2					451	
Sycamore Creek	1	379	223	185	92	19	4	3		10		3			899	
Total 1994	1319	2644	1810	776	428	171	23	1473	7	31	4	14	5		8705	

B Total number of fish collected at seven sites in upper Verde River, May 1995.

	Species															Total
	Longfin dace	Desert sucker	Sonora sucker	Roundtail chub	Spikedace	Speckled dace	Common carp	Red shiner	Fathead minnow	Yellow bullhead	Green sunfish	Smallmouth bass	Channel catfish	Flathead catfish		
BurntRanch		15	80	3	33			7		2		3			123	
FR638	11	48	57	4	2		1	8		3	8	1			143	
Duff Spring		65	73	50			1	55		17	5	8	2	1	275	
Bear Siding		45	47	22				10		5	10				139	
Perkinsville	1	90	10	115	1	23	1	4		1	3				249	
Railroad Bridge		36	38	43	19	2	1	8			1				148	
Sycamore		29	37	104	17		2	5		1	2				197	
Total 1995	12	328	322	341	72	25	8	97		29	29	10	2	1	1274	

C Total number of fish collected at seven sites in upper Verde River, April-May 1996.

Total number of fish collected at seven sites in upper Verde River, April-May 1996.															
	Species														
	Longfin dace	Desert sucker	Sonora sucker	Roundtail chub	Spikedace	Speckled dace	Common carp	Red shiner	Fathead minnow	Yellow bullhead	Green sunfish	Smallmouth bass	Channel catfish	Flathead catfish	Total
Burnt Ranch	91	79	92	23	33		1	88		1	1	5			414
FR638	179	127	307	81	53	1	5	112			1				868
Duff Spring	6	32	51	17			5	1		1		12			125
Bear Siding		50	25	6		1	1	27			2	10			125
Perkinsville	5	112	78	57		83		36		4		1		1	355
Railroad Bridge		33	82	50	3	3		2			2	4			159
Sycamore	1	38	41	25	51		1	9							166
Total 1996	282	471	654	259	140	88	13	275		6	6	32		1	2207

Table 1. Continued

D

Total number of fish collected at seven sites in upper Verde River, spring 1997.

Site	Longfin dace	Desert sucker	Sonora sucker	Roundtail chub	Spike- dace	Speckled dace	Yellow bullhead	Carp	Red shiner	Channel catfish	Mosquito- fish	Flathead catfish	Green sunfish	Smallmouth bass	Fathead minnow	Total
Burnt Ranch		88	88	24			13	3	1		3		4	8		189
FR838	18	78	57	8			10	1	82						1	254
Duff Spring	2	0	11	5			0	12	830	1					8	887
Bear Siding		18	4						8			1	3	11		48
Perkinsville		15	28			1	1		1222				1		1	1289
Black Bridge		44	34	13					77						2	170
Sycamore		12	38				18	3	17						8	92
	21	231	240	50		1	40	18	2238	1	3	1	8	35		2888

E

Total number of fish collected at seven sites in upper Verde River, spring 1998.

Site	Longfin dace	Desert sucker	Sonora sucker	Roundtail chub	Spike- dace	Speckled dace	Yellow bullhead	Carp	Red shiner	Channel catfish	Mosquito- fish	Flathead catfish	Green sunfish	Smallmouth bass	Fathead minnow	Total
Burnt Ranch			21	43	12				1	27		5		8	11	128
FR838	12		28	27	18				1	101		1		1	5	195
Duff Spring			10	18	23			8	2	78			1	2	18	181
Bear Siding			15	8	1			4	4	8				8	4	49
Perkinsville			35	10	5		12	5		701				1	3	772
Black Bridge			13	11	4			8		132				2	17	187
Sycamore		1	4	8				7	1	1					5	27
	13	128	124	84	0	12	12	33	8	1047	0	8	1	21	64	1520

Table 2. Relative abundance (%), number, and surface area of aquatic macrohabitats in time and space, upper Verde River, 1994-97.

Sample	Reach Year	Habitat type					Number	Surface area, m ²
		Glide	Run	LGR	HGR	POOL		
Burnt Ranch								
	1994	1	57	35	0	7	10	2124
	1995	78	0	9	0	13	14	1712
	1996	38	7	23	8	24	11	965
	1997	15	16	22	0	47	13	828
FR 638								
	1994	33	14	37	11	5	11	1411
	1995	48	30	9	11	3	12	2674
	1996	32	11	26	13	18	12	1450
	1997	21	26	29	10	15	18	1134
Bear Siding								
	1994	19	19	12	11	38	10	1486
	1995	16	0	27	9	48	13	5157
	1996	37	5	26	13	19	13	2231
	1997	51	4	5	12	29	14	1283
Duff Spring								
	1994	67	6	15	9	3	10	1913
	1995	36	8	7	3	47	13	2483
	1996	22	0	4	2	72	13	1554
	1997	12	15	8	10	55	16	2303
Perkinsville								
	1994	21	30	16	25	7	10	2279
	1995	36	10	10	12	33	11	3307
	1996	47	7	0	39	7	7	2150
	1997	47	8	26	11	9	14	969
Black Bridge								
	1994	33	37	30	10	0	7	1185
	1995	10	25	18	7	41	11	2390
	1996	23	19	0	23	35	11	1303
	1997	6	18	26	0	49	11	812
Sycamore								
	1994	0	40	27	21	11	10	2111
	1995	24	21	24	0	32	10	4970
	1996	42	29	0	22	6	8	1953
	1997	6	58	26	3	7	15	766

APPENDIX I

PUBLICATIONS, VERDE RIVER, 1995-97

PHYSICAL HABITAT USE BY SPIKEDACE IN THE UPPER VERDE RIVER, ARIZONA

Andrea P. Neary,¹ John N. Rinne² and Daniel G. Neary²

The spikedace, *Meda fulgida*, is a threatened species of short-lived minnow that persists only in the mid-elevations of the upper Gila River in New Mexico, and Aravaipa Creek and the upper Verde River in Arizona (Rinne 1991). The fish is mottled on the back, silvery on the sides, and whitened ventrally (Minckley 1973). Males become golden on their dorsal and lateral surfaces during breeding. Spawning is in the spring and early summer. One-year-old females produce one batch of 100 to 300 eggs, and two-year-olds may produce two in a single spawning period. Young emerge in April and May and usually grow to lengths of 35–40 mm by November. Individuals seldom exceed four years in age and 65 mm in length. Spikedace feed on aquatic and terrestrial insects as well as the fry of other species.

The upper Verde River is one of the most important habitats for the spikedace. It is a relatively wild, free-flowing stream that has low baseflows (0.57 m³/sec), no perennial tributaries, and periodic drought and extreme flood events (Stefferd and Rinne 1995). Although adapted to southwestern streams, spikedace populations are affected by natural variations in the Verde River hydrograph (i.e. alternating floods and drought). Habitat degradation produced by impoundment construction, dewatering of streams for irrigation, channel erosion, and sedimentation have contributed to the decline of spikedace populations. In addition, introduction of non-native fishes such as the red shiner has seriously affected the species through competition for food and habitat (Rinne 1991; Douglas et al. 1994). Finally, livestock grazing has been implicated in the decline of spikedace habitats and populations through streambank deterioration and sedimentation. However, the impact of grazing relative to non-native species predation is still unknown.

A major consideration in studying this rare species of native fish is the determination of its preferred habitat. Past surveys may have missed the spikedace because of sampling in the wrong microhabitats. Reconnaissance surveys of the upper Verde River completed in 1993 and 1994 indicated that within stream reaches actually occupied by spikedace, the fish were concentrated in discrete microhabitats.

This preliminary study had two major objectives. First, we planned to examine and define the preferred physical habitat of spikedace relative to water velocity, stream gradient, and stream substrate composition. Second, we wished to make specific sampling recommendations for future studies.

Methods

The site of the fish sampling is in the upper Verde River, Yavapai County, about 6 km downstream of Sullivan Lake. This reach of stream is known as the Burnt Ranch section (see Stefferud and Rinne 1995). It is subject to periodic drought and extreme flood events such as the 70+ year event which occurred in early 1993. Stream types within this reach include rocky, high-gradient riffles; lower gradient riffles consisting mainly of gravel; gentle runs with few large rocks, gravel, and some sand; and wide, gentle glides with mainly sandy bottoms (Rinne and Stefferud 1996).

Fish sampling and physical habitat characterization occurred on 14 different stream microhabitats within several reaches at the Burnt Ranch site. Each distinct microhabitat was classified into one of five types: high-gradient riffle (HGR), low-gradient riffle (LGR), run (RUN), glide (GLD), and pool (POL). Only two runs were found within the sampling areas, and pools were not evaluated based on previous information on the absence of spikedace in this microhabitat. This resulted in evaluation of 14 microhabitats. Each microhabitat was sampled using a 5 x 15 m seine with 3 mm mesh. Fish were collected by dragging the seine

¹Flagstaff High School, Flagstaff, AZ.

²USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Flagstaff AZ.

from up to downstream in microhabitats with pebble to sand bottoms. In microhabitats with extensive cobbles and boulders (i.e. high-gradient and low-gradient riffles), fish were collected by employing direct current, backpack electrofishing gear. In this approach, the seine was used to block the downstream end of a sample microhabitat, and the microhabitat was sampled from upstream to downstream into the net using the electrofishing unit. After either procedure, the seine was brought ashore and captured fish were immediately placed into a water-filled bucket. All the collected fish were identified by species and placed on a measuring board. Total lengths from snout to tail were recorded for use in other studies, and the fish were released back into the Verde at the site of capture.

After sampling the fish in the study area, data were collected to establish the velocities, depths, gradients, and substrates of each sampled microhabitat. Equipment and established procedures to determine these physical habitat measurements are described in Rinne and Stefferud (1996). Basically, stream velocities and depths were measured at three transects within each sample reach by a direct readout current meter attached to a 2-m rod with centimeter calibrations. When the depth was established and recorded, the bottom of the current meter was then held at 60 percent of the depth from the water surface to obtain mean velocity of the microhabitat. The velocity was recorded when the current meter readout came to a constant reading. Gradients and microhabitat dimensions were determined using a Criterion 400 laser survey instrument. Once into the survey measurements screen, the Criterion was aimed at a target with a reflector that sends back the laser beam. Each target was positioned at the upstream end of the sample site with the Criterion at the downstream end to determine gradient. It was also set up across the stream to determine width. When the Criterion laser fired, information such as distance, azimuth, percent slope, and gradient were automatically computed and presented on the data screen.

To characterize stream substrates, a modified pebble count technique developed by Bevenger and King (1995) was used. Using this method, one sample of the substrate, possibly consisting of sand, gravel, pebbles, cobbles, or boulders, was picked up and measured on the longest dimension with a ruler at the end of every three steps. Thirty measurements were taken throughout the entire sampling site in a random zig-zag pattern. A

simple average of the measurements gave the average sediment size for that site.

Results and Discussion

Spikedace microhabitat use was evident in results from stream sampling conducted in 1994 and 1995 (Figure 1). However, microhabitats were visually characterized during the 1994–1995 samplings, and general measurements of depth, velocity, and substrate material were made. No attempts were made in 1994–1995 to systematically classify microhabitats based on physical measurements. Some microhabitats that were identified visually as pools were actually mid-channel pools that were part of other microhabitat types. Since the ability to separate these microhabitats in a reliable manner was uncertain, plans were developed to add measurements of gradient, velocity, depth, and substrate to further refine the definition of the microhabitats.

Physical habitat information gathered during the March, 1996, survey for spikedace was compared to visual estimates of microhabitat type taken at each site. As a result of the laser survey measurements, each habitat was clearly associated with a range of gradients (Figure 2). Glides were less than 0.25 percent, runs were between 0.25 percent and 0.50 percent, low-gradient riffles were between 0.51 percent and 1.00 percent, and high-gradient riffles were greater than 1.00 percent. This gradient separation allows for a more consistent and accurate definition of habitat type than visual determinations.

Spikedace were found in greatest abundance in gradients between 0.4 and 0.6 percent (Figure 3). Accordingly, these gradients fall within runs to low-gradient riffle microhabitat classifications, as we have suggested. These two microhabitats contained almost 60 percent of the spikedace captured in the entire upper Verde River during 1994–1995 (Rinne and Stefferud 1996). Therefore, there is good agreement between the qualitatively defined habitats and those we have defined by laser surveying.

Spikedace were found in habitats with substrates of less than 10 percent sand (Figure 4). According to the greater density of spikedace present in stream reach 3, we determined that spikedace favored habitats with lower sand content. Glides, runs, and low-gradient riffle microhabitats contain increasingly less sand substrate, respectively (Rinne and Stefferud 1996). The data show that spikedace certainly prefer gravel over sand substrate during the spring spawning season. More

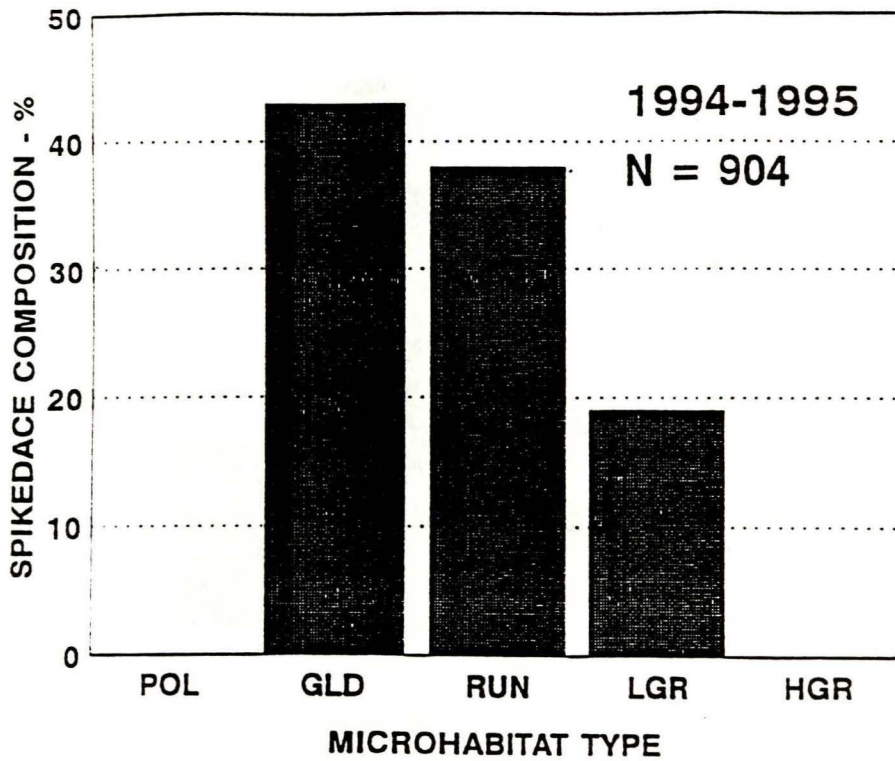


Figure 1. Spikedace distribution by microhabitat type, upper Verde River, Arizona (adapted from Rinne and Stefferud 1996).

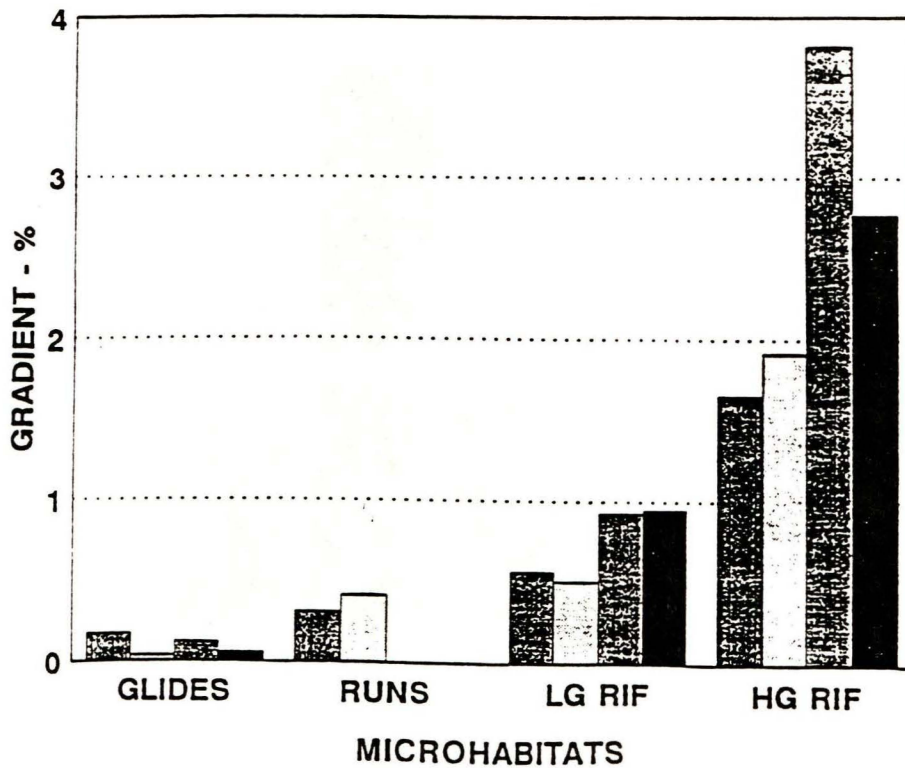


Figure 2. Comparison of gradients of 14 microhabitats, upper Verde River, March 1996.

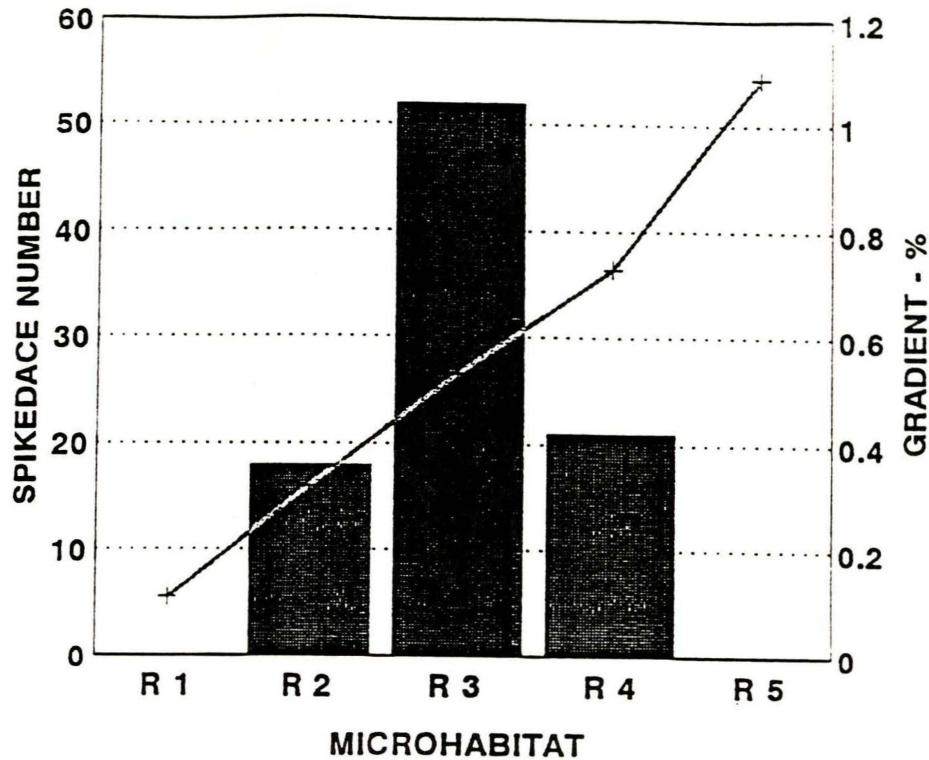


Figure 3. Spikedace numbers and stream gradient, Burnt Ranch section, upper Verde River, March 1996.

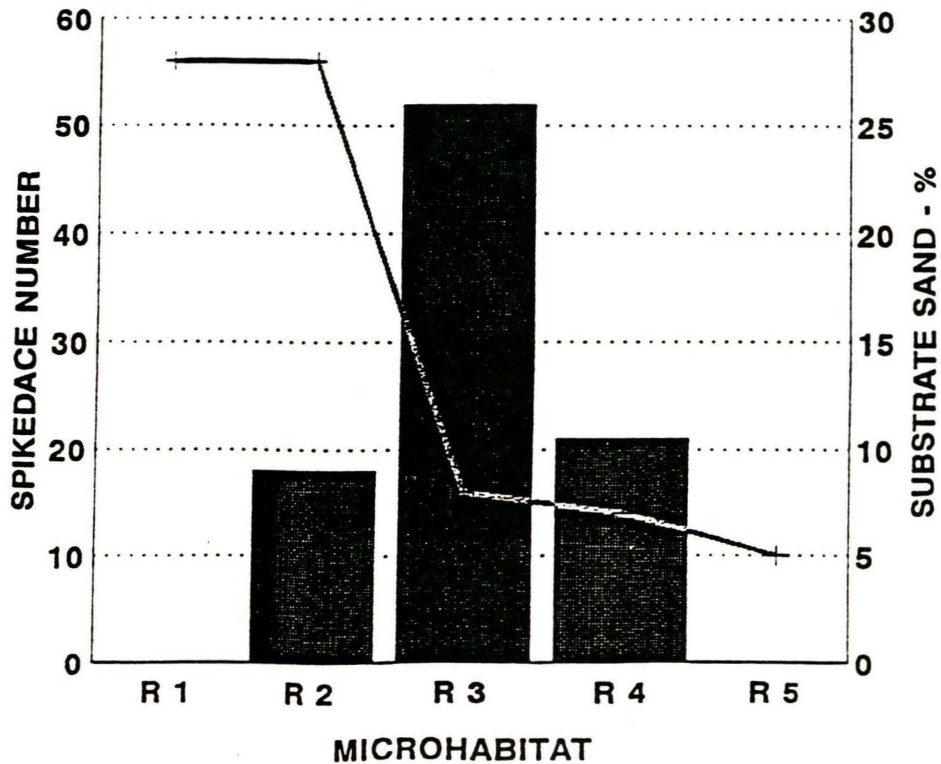


Figure 4. Spikedace numbers and percent sand in streambed substrate, Burnt Ranch study section, upper Verde River, March 1996.

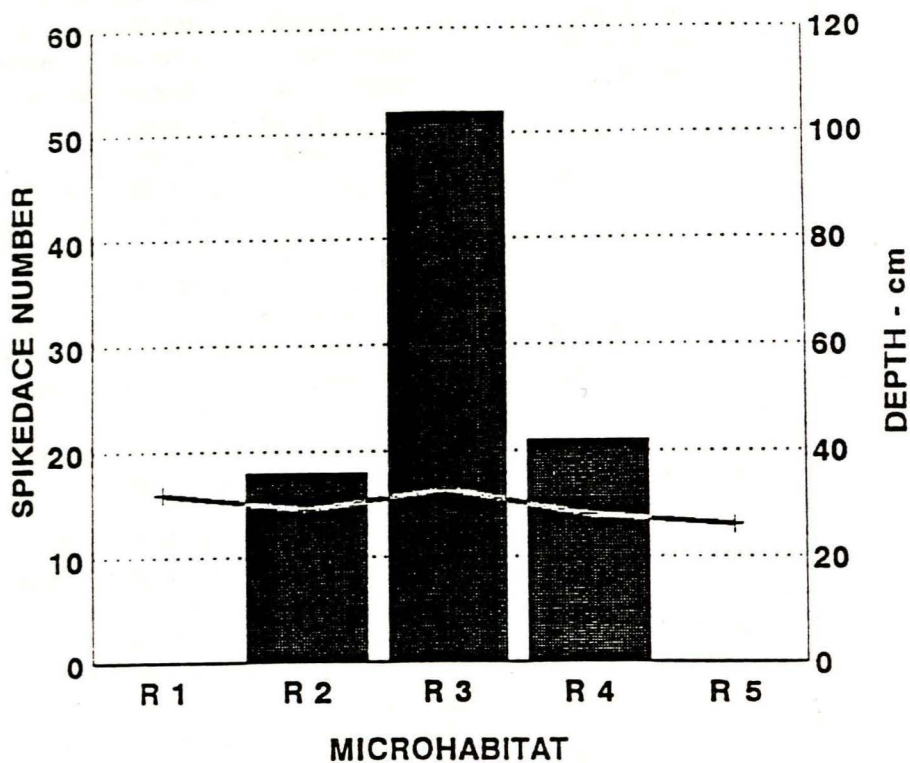


Figure 5. Spikedace numbers and stream depth, Burnt Ranch section, upper Verde River, March 1996.

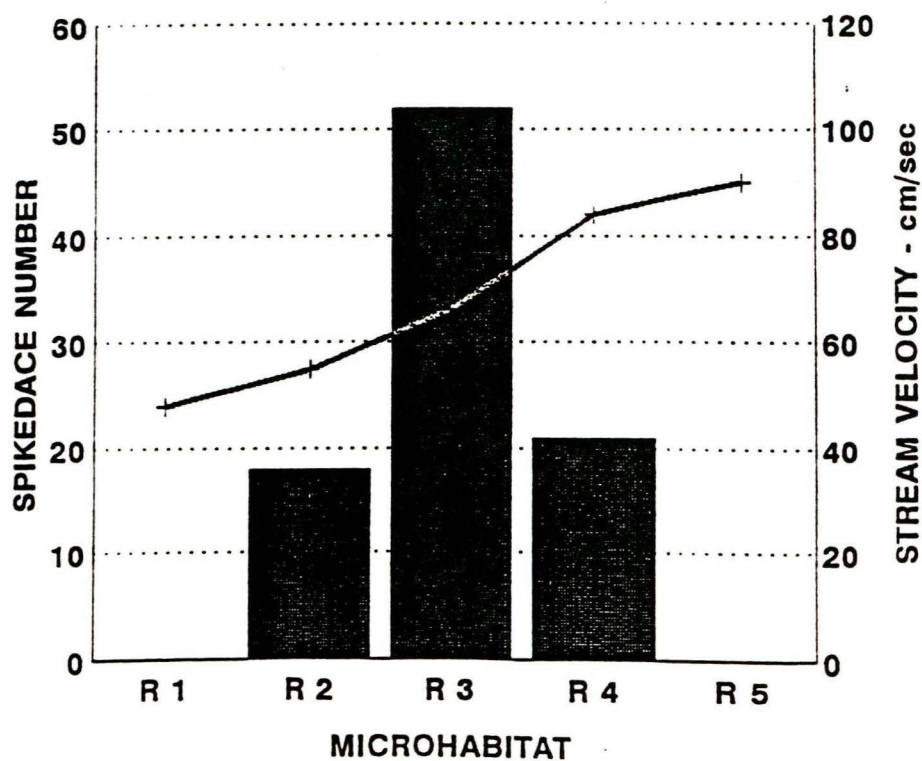


Figure 6. Spikedace numbers and stream average velocity, Burnt Ranch study section, upper Verde River, March 1996.

field sampling is needed to determine if these fish move into other microhabitats at different seasons.

Depth of water appeared to have little influence on the presence or absence of spikedace (Figure 5). That is, depth was approximately the same among all sample sites, yet only some areas contained spikedace. So factors other than depth appear to influence spikedace microhabitat use.

Velocity, however, was a greater determining factor in the presence or absence of spikedace. The species was found only in medium velocity (55–85 cm/sec) habitats. Where spikedace were absent, stream velocities averaged <50 cm/sec or >90 cm/sec (Figure 6). By comparison, where present, stream velocities averaged 65 cm/sec, which is characteristic of low-gradient riffles (Figure 2).

Conclusions and Recommendations

Briefly, our study brought us to specific conclusions as to the preferred habitats of the spikedace. Spikedace were found only in habitats with a specific range of velocity, substrate, and gradient characteristics. They were found in areas with velocities ranging from 55 to 85 cm/sec (mean 65), gradients of 0.4 to 0.6 percent (mean 0.5%) substrates of less than 10 percent sand, and in narrower channel sections.

For help in future sampling of the spikedace, all these factors should be taken into consideration and sampling should mostly take place in low-gradient riffles and runs. Sampling outside that range will greatly underestimate populations.

References Cited

- Bevenger, G.S. and R.M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. USDA Forest Service Research Paper RM-319, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 17 pp.
- Douglas, M.E., P.C. Marsh and W.L. Minckley. 1994. Indigenous fishes of western North America and the hypothesis of competitive displacement: *Meda fulgida* (Cyprinidae) as a case study. *Copeia* 1:9–19.
- Minckley, W.L. 1973. Fishes of Arizona. Arizona Game and Fish Department, Phoenix, AZ. 293 pp.
- Rinne, J.N. 1991. Habitat use by spikedace, *Meda fulgida* (Pisces: Cyprinidae), in southwestern streams with reference to probable habitat competition by red shiner, *Notropis lutrensis* (Pisces: Cyprinidae). *The Southwestern Naturalist* 36:7–13.
- Rinne, J.N. and J.A. Stefferud. 1996. Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona. In *Hydrology and Water Resources in Arizona and the Southwest*, Vol. 26. Proceedings of the 1996 Meetings of the Arizona Section, American Water Resources Association and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 20, 1996, Tucson, Arizona.
- Stefferud J. A. and J. N. Rinne. 1995. Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes. Pages 26–32 in *Hydrology and Water Resources in Arizona and the Southwest*, Vols. 22–25. Proceedings of the 1995 Meetings of the Arizona Section, American Water Resources Association and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 22, 1995, Flagstaff, AZ.

BASEFLOW TRENDS IN THE UPPER VERDE RIVER RELATIVE TO FISH HABITAT REQUIREMENTS

Daniel G. Neary and John N. Rinne¹

Although much attention has been given to the effects of stormflows on native fish in Arizona's rivers (Minckley 1973, Rinne and Minckley 1991), it is minimum baseflows that are most critical for fish survival. Concerns have been raised about the potentially negative effects of drought, consumptive water use, watershed management, and water diversion on native fish populations, but significantly reduced baseflows could also put these fish populations at risk. Because of the controversy over threatened and endangered fish such as the spinedace (*Medeo fulgida*) in the upper Verde River, it is important to examine recent trends in minimum baseflows on this river.

Because of the vagaries of climate (Green and Sellers 1964), characteristic to the Basin and Range physiography of the Southwest (Nations and Stump 1981), and its aridity (Jaeger 1957, Dunbar 1968), patterns of streamflow are highly variable. An alternating pattern of floods and drought are the norm. The interactions of these factors result in definition of aquatic habitat or the lack of it in the region (Rinne 1995a). Because of the Southwest's climate pattern and frequent water shortages, damming of major rivers was extensive during the first 60 years of this century. Commencing with the completion of Roosevelt Dam, the first Bureau of Reclamation dam on the Salt River basin of Arizona in 1911, modification of rivers in this region and throughout the West has been widespread. Rinne (1994) reported that 70 percent of the main-stream rivers in Arizona have been modified by dams. Such modifications combined with rapid growth in population and economic activities have resulted in loss of surface flow and changes in the quantity and quality of streamflow across the landscape. Free flowing, un-dammed rivers are a rarity in this region.

The upper Verde River is one of the few remaining reaches of wild, free-flowing rivers in Ari-

zona. Furthermore, this reach of river is a rarity in the Southwest because it supports a native fish community (Stefferd and Rinne 1995). Studies of the reach of river upstream from Sycamore Creek began in 1994 following major flooding events on the Verde in the winter of 1992-93. The main study objective is to examine the sustainability of the native fish fauna relative to abiotic and biotic factors, specifically the role of the changing hydrograph with time and space, and the effect of introduced fishes on the sustainability of the native fish fauna (Stefferd and Rinne 1995, Rinne and Stefferud 1996). Other components of the overall study effort are the definition of aquatic macrohabitats and their utilization by fishes (Rinne and Stefferud 1996, Neary et al. 1996, Rinne and Neary 1997, Sponholtz and Rinne 1998). Southwestern fishes are highly adapted to the varying cycles of water feast and famine (Minckley 1973, Deacon and Minckley 1974, Rinne and Minckley 1991). However, they are not able to adapt to the complete loss of surface flow, the obvious, critical component to their survival (Rinne 1995b).

Examination of baseflow hydrology in the upper Verde River basin was undertaken in the study reported here to determine the trends in minimum baseflows over the past three decades. Because of recent droughts and the extensive, rapidly increasing urban development in Chino Valley, the headwater of the Verde River drainage system, concerns have arisen over the ability of the river to sustain its native fish population. Understanding this trend is very important for resource managers, who must arrive at decisions regarding land use activities and sustainability of the Verde River spinedace and other native fish. This paper delineates the trends in the upper Verde River minimum mean daily flow, suggests probable mechanisms for the observed changes, and emphasizes the importance and significance of changing minimum flows to the native fish fauna.

¹ Rocky Mountain Research Station, USDA Forest Service, Flagstaff, AZ.

Methods

Streamflow records were obtained from the U.S. Geological Survey, Water Resources Division, Arizona District for the Paulden gage (number 9503700) from 1963 through 1995, the Clarkdale gage (number 9504000) from 1965 through 1995, and the Camp Verde gage (number 9506000). The Paulden gage is located on the Verde River (perennial flow) between the confluence of Granite Creek (ephemeral flow) and Hell Canyon (ephemeral flow), about 14 km below the beginning of perennial streamflow. Its contributing drainage is about 5,568 km², or 40 percent of the Verde's 14,000 km² basin area (House et al. 1995). The Clarkdale gage is located below the Verde's confluence with Sycamore Creek, about 42 km downstream of the Paulden gage. Its flow-contributing area is 8,148 km², or 60 percent of the total river drainage basin. The Camp Verde gage is located downstream of Camp Verde, with a drainage area of 12,028 km², or 85% of the total basin area. Flow data are available on the Verde River Watershed Association web site at <http://www.verde.org>. Monthly and annual discharge data were taken from U.S. Geological Survey gage summary tables. Mean daily flows were analyzed to determine the minimum mean daily flow on an annual basis.

Precipitation data were taken from the Prescott precipitation station (number 026796) for the period 1931 through 1994. The rain gage is about 46 km south of the Paulden streamflow gage.

This preliminary analysis investigates trends in water flow data to guide a more complete analysis of water flow parameters at a later date. It considers only annual minimum daily baseflows and annual maximum daily peakflows.

Baseflow Trends

Mean monthly discharges at Paulden range from 0.67 to 3.62 m³/s. The mean maximum discharge of 40.77 m³/s normally occurs in February, and the mean minimum discharge of 0.45 m³/s occurs in May. During the period of record (1963–1995), the maximum mean daily flow was 387.85 m³/s in 1993. An instantaneous peakflow of 657 m³/s occurred during the 20 February 1993 storm (House et al. 1995). The minimum mean daily flow was 0.42 m³/s in 1964.

The annual minimum daily flows for the period of record that represent the lowest baseflow during each year are shown in Figure 1. They range from a low of 0.42 m³/s in 1964 to a high of 0.71 m³/s in 1985, 1986, 1994, and 1995. There is a trend of increasing annual minimum daily flows over

this period. The 5-year running average rose from 0.52 m³/s in 1967 to 0.67 m³/s in 1995, a 29 percent increase. As of 1995, there does not appear to be any evidence to indicate that water use in the upper Verde River watershed over the past three decades has affected the annual minimum daily flow.

Mean monthly discharges at Clarkdale range from 1.81 to 14.64 m³/s. The mean maximum discharge of 98.80 m³/s normally occurs in February, and the mean minimum discharge of 1.76 m³/s occurs in June. During the period of record (1965–1995), the maximum mean daily flow was 849.30 m³/s in 1993. An instantaneous peakflow of 1,507 m³/s occurred during the 20 February 1993 storm (House et al. 1995). The annual minimum mean daily flow was 1.70 m³/s in 1978.

The annual minimum daily flows at the Clarkdale gage for the period of record are shown in Figure 2. They range from a low of 1.81 m³/s in 1966, 1967, 1972, and 1981 to a high of 2.32 m³/s in 1993. Similar to the trend higher up the Verde River at Paulden, there is also a trend of increasing annual minimum daily flows over this period. The 5-year running average rose from 1.88 m³/s in 1969 to 2.18 m³/s in 1995, a 16 percent increase.

As streamflow gaging at Paulden and Clarkdale did not start until 1963, it is difficult to determine where the current trends in minimum daily baseflow fit in the long-term pattern of minimum daily baseflow. Data from the incomplete streamflow record at Camp Verde (Figure 3) indicate that annual minimum daily flows in recent times (1989–1994) have been less than those during the 1934–1945 period. It is also interesting to note that the annual minimum daily baseflow decreases between the Clarkdale gage and the Camp Verde gage. This decline is indicative of not only the water diversions that occur in the Clarkdale–Camp Verde reach, but also water surface evaporation, transpiration by riparian vegetation, and seepage into the alluvium of the river.

Mechanisms

The trend of increasing annual minimum daily flows at both the Paulden and Clarkdale stream gaging sites suggests that baseflows in the upper Verde River watershed are increasing in response to either increases in precipitation or reductions in evapotranspirational losses. There is no evidence of the latter, and no major vegetation management programs have occurred in the past three decades that would reduce transpirational losses of water. The remaining mechanism that might explain

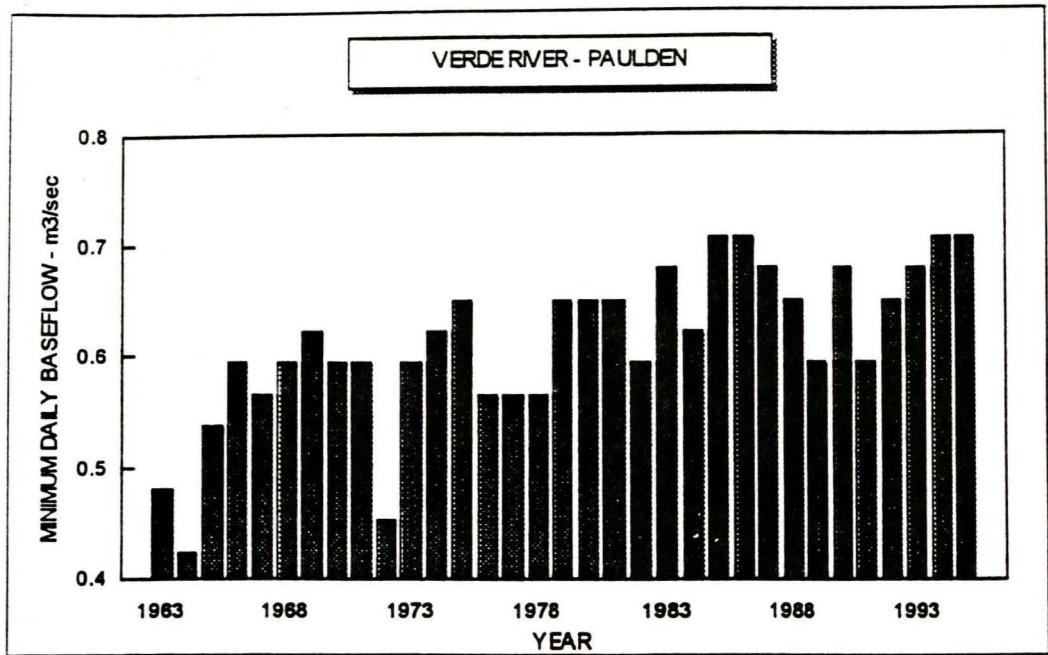


Figure 1. Annual minimum daily baseflows on the Verde River at the Paulden streamflow gage (1963–1995).

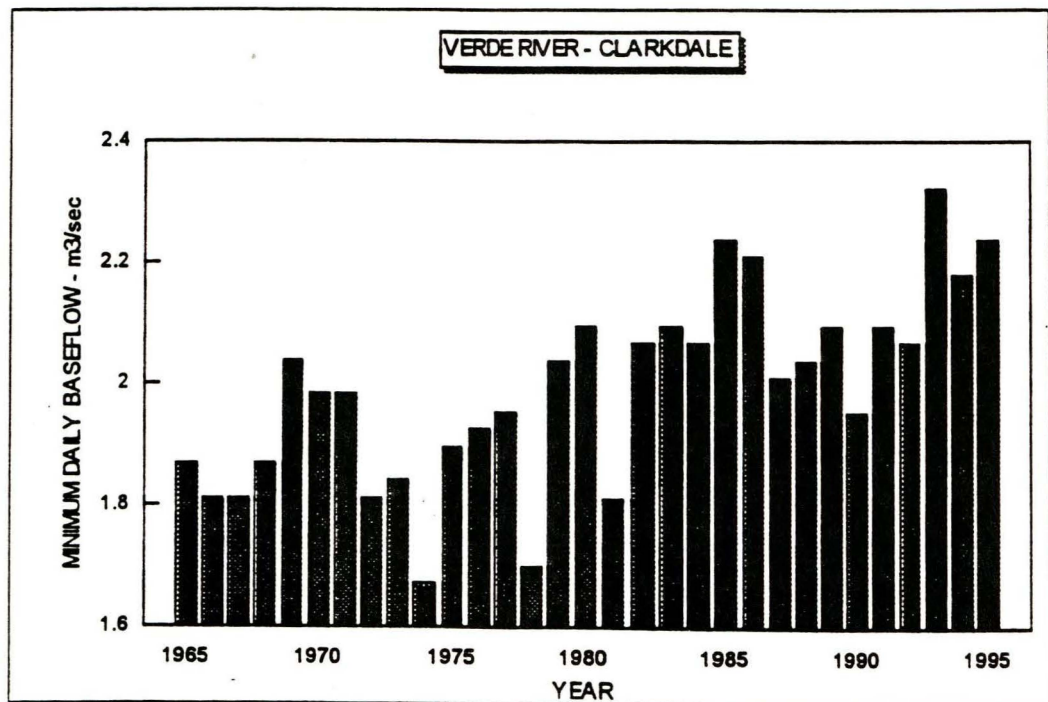


Figure 2. Annual minimum daily baseflows on the Verde River at the Clarkdale streamflow gage (1965–1995).

increases in minimum daily baseflows is precipitation.

Figure 4 shows rainfall at Prescott as a departure from the 1932–1994 mean of 1933 mm. During this time frame there have been wetter and drier periods as indicated by the cumulative departure curve (e.g. wetter, 1932–1946; drier, 1946–1964; wetter, 1965–1972; drier, 1973–1981; wetter, 1981–1988; drier, 1989–1994). The 1964–1994 segment shows a general wetting trend during the period of increasing minimum daily flows in that cumulative departures from the mean have become more positive since the –2,180 mm low point in 1964. This indicates that annual minimum daily flows still seem to generally respond to oscillations in precipitation and not some other hydrologic process such as evapotranspiration. A downward trend in vegetation density in the watershed could possibly reduce evapotranspiration and increase baseflows. Based on photographic evidence, vegetative cover in the upper Verde River riparian zone over the past 30 years has been increasing, not decreasing. There is not enough evidence on upland areas to argue one way or the other.

During the period of time in which annual minimum daily baseflows at the Paulden and Clarkdale gages have been increasing, annual maximum daily peakflows have also been increasing (Figures 5 and 6). Since peakflows have a high degree of year-to-year variability, they are more indicators of an increasingly wetter climate than controlling factors in rising minimum daily flow. Peakflows have definitely increased over the period of record for both the Paulden and Clarkdale gages, culminating in the high flows of February 1993 that were estimated to have a return period of 70 years.

Significance to Fishes

Miller (1961) first reported the decline of native southwestern fishes as a result of man's activities. Dam construction, diversion, and groundwater mining were listed as major factors resulting in aquatic habitat alteration and loss in the Southwest. Rinne and Minckley (1991) further emphasized the continuing decline of the native desert fishes in the region. The upper Verde has been largely spared from the first two major impacts; the influence of the last, groundwater pumping, may be yet to come. Nevertheless, the recorded increase in baseflow on the upper Verde is not only surprising but certainly beneficial to the native fish fauna. Just as floods have been documented to be beneficial to native fishes in this reach of river (Rinne and Stefferud 1996), base-

flows are critical to sustaining the native fish community in times of reduced precipitation and runoff. The quantity and quality of streamflow, in time and space, strongly dictate fish habitat and in turn fish populations.

Although unequivocal, quantitative definitions of aquatic macrohabitats in the upper Verde are yet in process (Sponholtz and Rinne 1998), utilization of riffles versus pool habitats has been relatively well defined for members of the native fish community inhabiting the upper Verde River (Rinne 1989, 1991, 1992, Rinne and Stefferud 1996). Shallower water, riffle-dwelling species such as loach minnow (*Rhinichthys cobitis*), spikedace (*Meda fulgida*), speckled dace (*R. osculus*), and longfin dace (*Agosia chrysogaster*) would be affected first if streamflow is decreased. Speckled dace, longfin dace and spikedace, a threatened species, currently inhabit the upper Verde (Stefferud and Rinne 1995). All three species have been demonstrated to inhabit low-gradient riffles (Rinne 1992, Neary et al. 1996, Rinne and Stefferud 1996) and would be the first to be negatively impacted if streamflow is decreased. Initially, a decrease in flow would reduce reproductive potential in riffles and ultimately would force these two species into pools inhabited by the native predatory species, roundtail chub (*Gila robusta*; Rinne 1992), and introduced smallmouth bass (*Micropterus dolomieu*) and yellow bullhead (*Ictalurus nebulosus*; Rinne and Neary 1997). Fortunately, the increase in baseflow can only be considered as a critical, positive effect on populations of these two species.

Conclusions and Recommendations

At the USGS Paulden gage, near the beginning of perennial flow on the Verde River, mean daily minimums range from 0.42 to 0.71 m³/sec. The annual minimum mean daily flow at this gage exhibits a trend of increasing minimum flows over the past three decades. At the Clarkdale gaging station above Cottonwood, mean daily minimums vary from 1.70 to 2.32 m³/sec. Similarly to Paulden, the trend in annual minimum mean daily flows at Clarkdale since 1965 is towards increasing annual minimum daily flows. At the present time, these increases in minimum baseflows on the upper Verde River appear to indicate that adequate flows will be available in the near future to sustain the Verde's fish population. This trend will require future evaluation and monitoring to determine if rapid urbanization of the Prescott and Chino Valley areas in future years will impact baseflow of the upper Verde River.

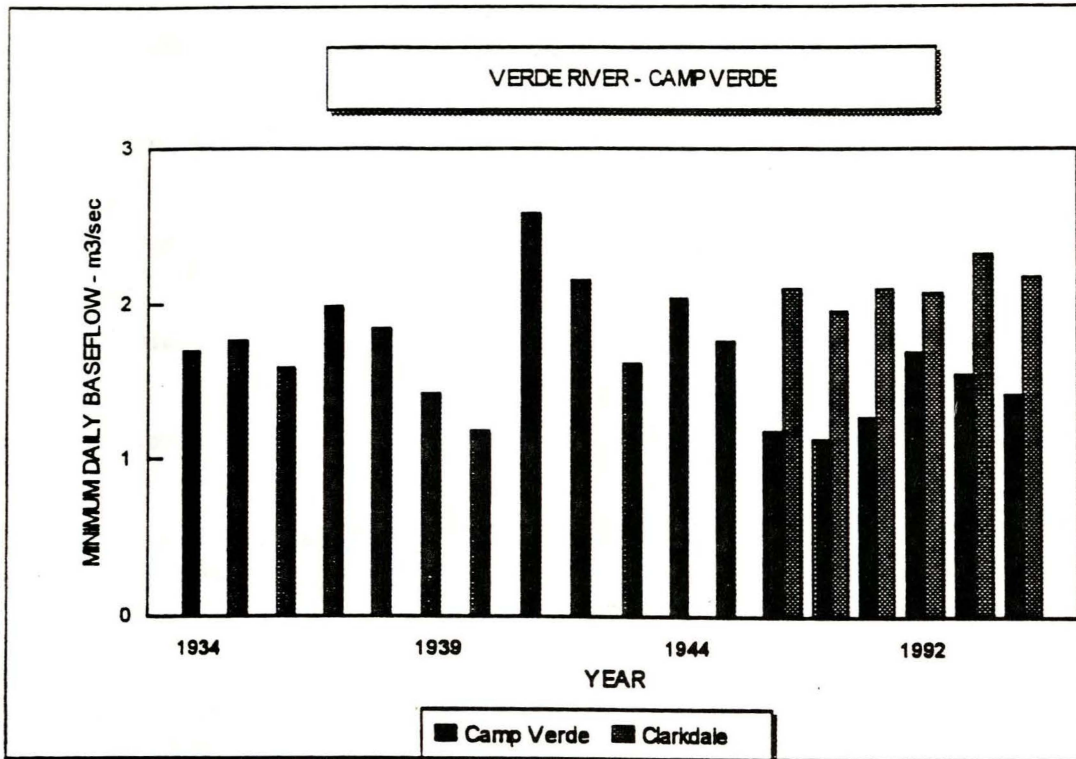


Figure 3. Annual minimum daily baseflows on the Verde River at the Camp Verde stream-flow gage (1934–1945 and 1989–1994).

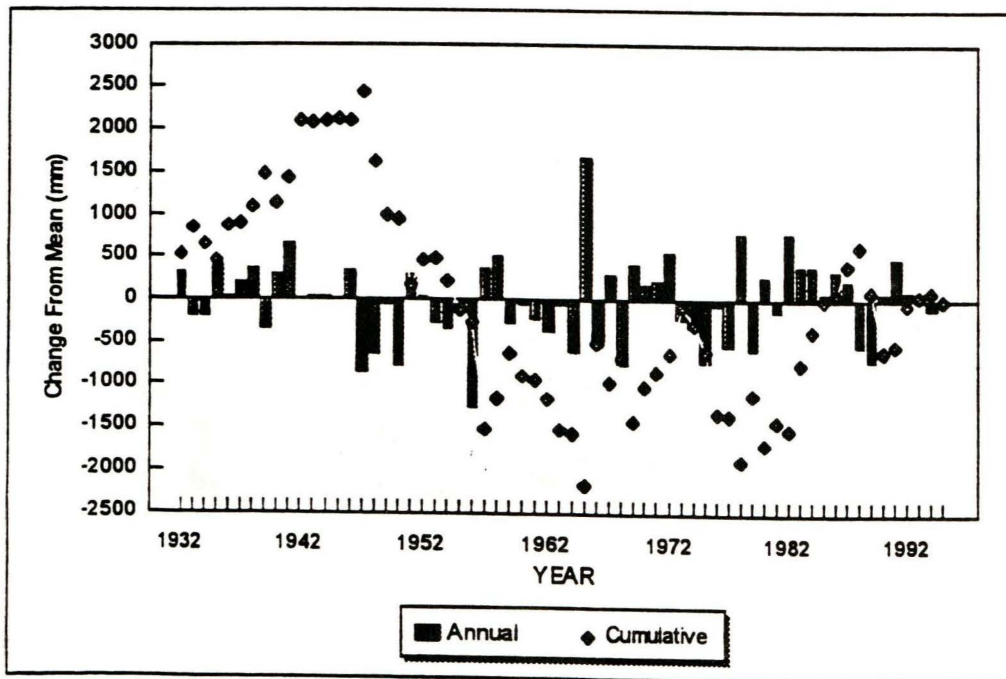


Figure 4. Annual and cumulative rainfall departure from the 1932–1994 mean, Prescott, Arizona, precipitation gage.

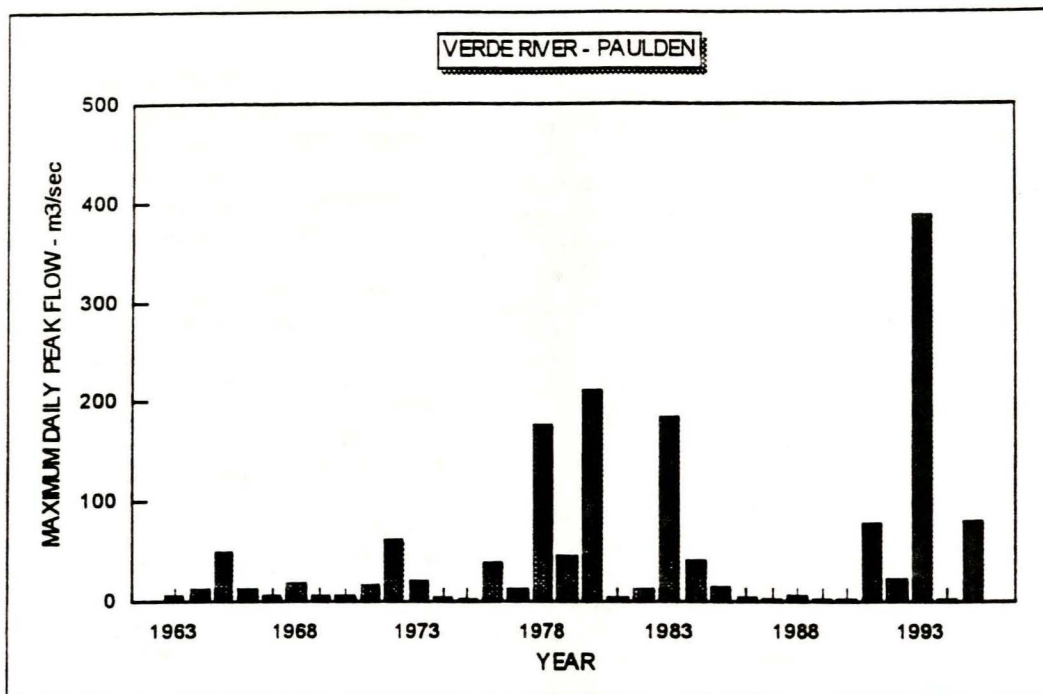


Figure 5. Annual maximum daily peakflows on the Verde River at the Paulden gage (1963–1995).

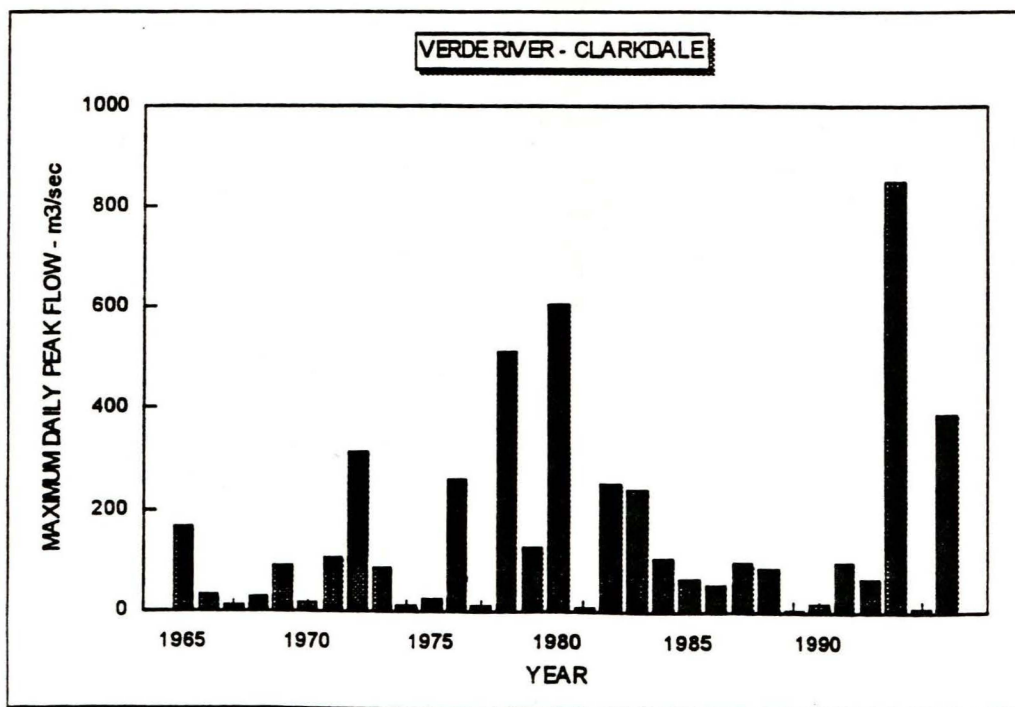


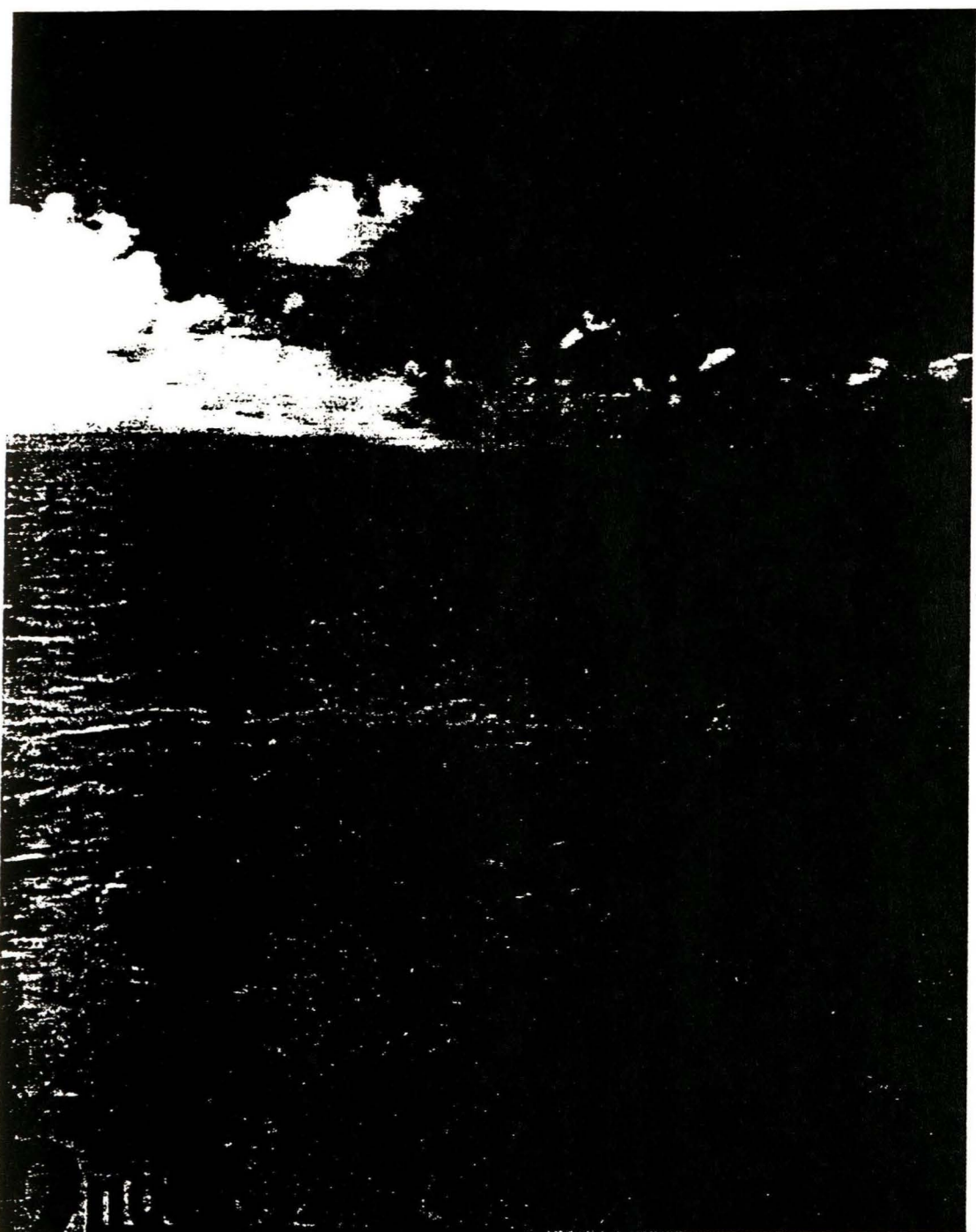
Figure 6. Annual maximum daily peakflows on the Verde River at the Clarkdale gage (1965–1995).

References Cited

- Deacon, J.E. and W.L. Minckley. 1974. Desert fishes. In G.W. Brown, Jr., editor, pp. 385-488. Desert biology, Volume 2. Academic Press, New York.
- Dunbier, R. 1968. The Sonoran Desert. Its geography, economy, and people. University of Arizona Press, Tucson.
- Green, C.R. and W.D. Sellers, editors. 1964. Arizona climate. University of Arizona Press, Tucson.
- House, P.K., P.A. Pearthree, and J.E. Fuller. 1995. Hydrologic and paleohydrologic assessment of the 1993 floods on the Verde River, Central Arizona. Arizona Geological Survey Open File Report 95-20, 23 pp.
- Jaeger, E.C. 1957. The North American deserts. Stanford University Press, Palo Alto, CA.
- Miller, R.R. 1961. Man and the changing fish fauna of the American Southwest. Papers of the Michigan Academy of Arts, Sciences and Letters 46: 365-404.
- Minckley, W.L. 1973. Fishes of Arizona. Arizona Game and Fish Department, Phoenix, 293 pp.
- Nations, D. and E. Stump. 1981. Geology of Arizona. Kendall/Hunt Publishers, Dubuque, IA, 221 pp.
- Neary, A.P., J.N. Rinne, and D.G. Neary. 1996. Physical habitat use by spinedace in the upper Verde River, Arizona. Hydrology and Water Resources in Arizona and the Southwest 26: 23-28.
- Rinne, J.N. 1989. Physical habitat use by loach minnow, *Tiagra cobitis* (Pisces: Cyprinidae), in southwestern desert streams. Southwestern Naturalist 34(1): 109-117.
- Rinne, J.N. 1991. Physical habitat use by spinedace, *Meda fulgida* in southwestern desert streams with probable habitat competition by red shiner, *Notropis lutrensis* (Pisces: Cyprinidae). Southwestern Naturalist 36(1): 7-13.
- Rinne, J.N. 1992. Physical habitat utilization of fish in a Sonoran Desert stream, Arizona, southwestern United States. Ecology of Freshwater Fishes 1: 1-8.
- Rinne, J.N. 1994. Declining southwestern aquatic habitats and fishes: Are they sustainable? In W.W. Covington and L.F. DeBano, technical coordinators, pp. 256-165. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report RM-247. Fort Collins, CO.
- Rinne, J. N. 1995a. Sky Island aquatic resources: Habitats and refugia for native fishes. In L.F. DeBano, P.F. Ffolliott, A. Ortega-Rubio, G.J. Gottfried, R.H. Hamre, and C.B. Edminster, technical coordinators, pp. 251-260. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report RM-264. Fort Collins, CO.
- Rinne, J.N. 1995b. Desired future condition: Fish habitat in southwestern riparian habitats. In D.W. Shaw and D.M. Finch, technical coordinators, pp. 336-345. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report RM-272. Fort Collins, CO.
- Rinne, J.N. and W.L. Minckley. 1991. Native fishes of arid lands: A dwindling natural resource of the desert Southwest. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report RM-206. Fort Collins, CO, 45 pp.
- Rinne, J.N. and J.A. Stefferud. 1996. Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona. Hydrology and Water Resources in Arizona and the Southwest 26: 13-22.
- Rinne, J.N. and D.G. Neary. 1997. Stream channel and fish relationships: Preliminary observations, Verde River, Arizona. In AWRA/ICOWR Symposium, Water Resources Education, Training, and Practice: Opportunities for the Next Century, pp. 475-482. Keystone, CO, June 29-July 3, 1997.
- Sponholtz, P. and J.N. Rinne. 1998 (this volume). Refinement of definition of aquatic macrohabitat definition in the upper Verde River, Arizona. Hydrology and Water Resources in Arizona and the Southwest 27.
- Stefferud J.A. and J.N. Rinne. 1995. Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes. Hydrology and Water Resources in Arizona and the Southwest 22-25: 26-32.

DEVELOPING AND SUSTAINING WORLD FISHERIES RESOURCES

The State of Science and Management



Editors
DA Hancock
DC Smith
A Grant
JP Beumer

World Fisheries

C O N G R E S S



FACTORS CONTRIBUTING TO COLLAPSE YET MAINTENANCE OF A NATIVE FISH COMMUNITY IN THE DESERT SOUTHWEST (USA)

John N. Rinne^A and Jerome A. Stefferud^B

^A USDA Forest Service, Rocky Mountain Station, The Southwest Forest Science Complex 2500 S. Pineknoll Drive, Flagstaff, Arizona 86001, USA.

^B USDA Forest Service, Tonto National Forest 2324 E. McDowell Road Phoenix, Arizona 85010, USA.

Summary

Data presented indicate that the ratio of population numbers of native to non-native fish species in the Verde River, Arizona fish community is variable but most consistently 85:15. In part, the relative numbers of each component are dependent on river location and species of fish. Characteristic stream hydrography, in terms of location, season, and severity and duration of flood events, also appears to be a contributing factor for maintaining this ratio. Although a 7-year flood event collapsed populations of both native and non-native species, the proportions of the two groups changed only slightly (82:18 to 85:15). By comparison, a 50+ year flood event resulted in native species increasing by 11% (86–97%). Reproduction and recruitment of natives is enhanced post-flood, apparently as a result of cleaning and rejuvenation of stream substrates. Based on a decade of data (1986 to 1996), the natural hydrography of the upper Verde River, as opposed to biological or native/non-native species interactions, is suggested as the controlling factor in sustaining native fishes in this large, low desert river.

INTRODUCTION

Patterns of stream discharge and the presence of introduced fishes have been suggested to interact to influence native fishes and their sustainability in streams of the arid American southwest (Minckley and Meffe 1987; Rinne and Minckley 1991; Rinne 1993, 1995a; and Stefferud and Rinne 1995). Commencing with the water reclamation era in the early 1900s, dams have been ubiquitously imposed on most rivers in this region (Rinne 1991, 1993, 1995b), drastically and irreversibly altering aquatic habitats (Miller 1960; Rinne and Minckley 1991). Non-native fish introductions commenced before 1900

and have steadily escalated (Rinne 1991, 1993, 1995a). The combination of these two impacts has markedly reduced both the range and numbers of all native species of fishes, rendering some extinct (Minckley and Douglas 1991), yet native fishes persist in most drainage basins in the Southwest. However, where both altered, unnatural stream hydrography and non-native fish exist, native fish generally are either reduced from historic levels or are absent (Minckley and Deacon 1968).

In all but a few streams, these two factors, when combined, have eliminated native fish species and modified community structure and dynamics, often within a few years after the action.

Streams where the native fauna remains predominant are a rarity in the Southwest. The exact mechanism that sustains native species in the presence of non-native species is not understood, but is hypothesized to be related to cycles of extreme droughts and floods that characterize streams in this region. Because of the ever-increasing demand for water for a growing populace, and the generally imperilled status of the southwestern native fish fauna (Rinne and Minckley 1991), clarification of this question is important at this time.

The upper Verde River, central Arizona, from its source at Sullivan Lake to the mouth of Sycamore Creek 60 km downstream is a rarity among southwestern rivers. It is the only reach of large ($1.3\text{--}7.0\text{ m}^3\text{ sec}^{-1}$ annual mean discharge) perennial desert river in Arizona where surface flows are not altered by dams or diversions. Further, this reach of river retains a native fish fauna that is largely intact. Six of the eight native species (desert sucker, *Catostomus clarkii*; Sonora sucker, *Catostomus insignis*; roundtail chub, *Gila robusta*; longfin dace, *Agosia chrysogaster*; spikedace, *Meda fulgida* and speckled dace, *Rhinichthys osculus*) are present in viable numbers (Stefferd and Rinne 1995); all are either state or federally-listed species. Two species (Colorado squawfish, *Ptychocheilus lucius* and razorback sucker, *Xyrauchen texanus*), which have been documented in archaeological sites along the Verde (Minckley and Alger 1968) but are no longer to be found, are currently objects of re-introduction and restoration efforts. Nevertheless, an assemblage of non-native fish occurs in this reach of river, including red shiner (*Cyprinella lutrensis*), fathead minnow (*Pimephales promelas*), common carp (*Cyprinus carpio*), green sunfish (*Lepomis cyanellus*), smallmouth bass (*Micropterus dolomieu*), yellow bullhead (*Amerius natalis*), and channel catfish (*Ictalurus punctatus*). Reasons for complete retention of the native fish are unclear. Whether the native species are maintaining their populations in the presence of non-native species, or whether the non-native species are unable to become dominant due to hydrologic conditions (water quantity and quality) and stream gradient is not known. Few published data are available to document either the history of encroachment by non-native species or trends in the dynamics of southwestern fish communities.

Although a number of studies on fishes and their habitats have been undertaken in the Verde River, none have investigated the long-term interrelationships of the native and non-native fish communities relative to flood disturbance events and to each other. The relationship of base and flood flows to sustainability of native fishes in the upper Verde River needs to be determined in order to provide input to water management plans for the system. The headwater Big Chino Valley aquifer is a major source of baseflow for the Verde River above Sycamore Creek; and there are proposals to extract its water to supply municipal needs to towns in the watershed. Depletion of discharge from the aquifer could alter baseflow in the river and affect sustainability of native fish. Other proposals to divert water directly from the Verde River and construction of mainstream dams have been made. However, the unresolved question of their effects on the native fish has deferred approval by resource agencies. Continued growth in the region will probably increase

pressure for water development and diversion from the upper Verde. Resource management agencies need to have reliable data in time and space to chart a course of action that will conserve native fish in the river.

In Spring 1994, we initiated a study to define the roles and relative influence of physical (hydrology, geomorphology) and biological (non-native fish population and community dynamics) factors on the sustainability of native fishes in time (10 years) and space (seven localities over 60 km of river) in this reach of free-flowing river. Initiation of the study immediately followed a 50+ year flood event in this reach of river. Although only in the third year of study, the intent of this paper is to provide baseline data and to discuss the role of these two factors in the collapse, re-establishment, and sustainability of the native fish community in a large, perennially-flowing desert river.

METHODS

Study sites were located primarily on reasonable access and linear disposition over the 60 km reach. Specific location was based on channel morphology and types of habitat present (Stefferd and Rinne 1995; Rinne and Stefferud 1996). Seven sites were chosen commencing at a point 7.6 km east of Sullivan Lake and ending at the confluence of Sycamore Creek (Fig. 1). Sites were linearly separated by distances of 3.3–16.2 km.

Initially, we proposed to sample 250–300 m of stream at each site; however, exact length was dependent on habitat complexity at the site. In general, a riffle-pool sequence was selected, then the study site was expanded to include other significant habitat types present (Rinne and Stefferud in press). Habitat types were sampled for fish with backpack DC electro-fish units, seines (3.2 mm mesh), dip-nets, and trammel nets progressing from downstream to upstream. All fish captured within a habitat type were identified to species, enumerated and returned alive to the stream. All sites were sampled in Spring; however, only the upper- (Burnt Ranch) and lower-most (Sycamore) sites were sampled in Autumn. Reduced (seven to two) sampling protocol

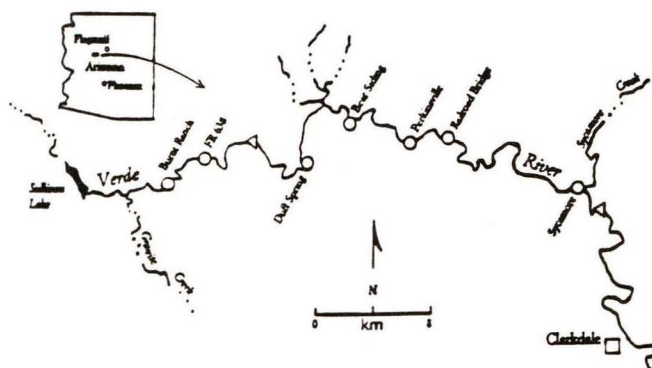


Fig. 1. The upper Verde River indicating the seven sites sampled between 1993 and 1996. Triangles indicate location of the Paulden and Clarkdale stream gauges.

Table 1. Fish community dynamics in the upper Verde River, Spring 1994–96. Values in () and [] are percent change from previous Spring, and bold numbers are native : non-native proportions. Species abbreviations are shown and are the same for Tables 2 and 3

Species	Spring 1994	Spring 1995 (%)	Spring 1996 (%)
Native species			
Desert sucker (CACL)	2644	247 (-90)	471 (+90)
Sonoran sucker (CAIN)	1810	322 (-82)	654 (+103)
Roundtail chub (GIRO)	776	341 (-56)	259 (-24)
Longfin dace (AGCH)	1319	12 (-99)	282 (+2250)
Spikedace (MEFU)	428	72 (-83)	140 (+94)
Speckled dace (RHOS)	171	25 (-85)	68 (+172)
Total (%) [%change]	7148 (82)	1019 (85) [-86]	1874 (85) [+714]
Introduced Species			
Red shiner (CYLU)	1473	97 (-93)	275 (+183)
Common carp (CYCA)	23	6 (-74)	13 (+117)
Smallmouth bass (MIDO)	14	10 (-28)	32 (+220)
Green sunfish (LECY)	5	29 (+480)	6 (-79)
Yellow bullhead (AMNA)	36	32 (-11)	6 (-84)
Fathead minnow (PIPR)	6	0 (-100)	0 (—)
Total (%) [% change]	1558 (18)	174 (15) [-89]	332 (15) [+71]

was based on both consideration of impact to fishes and to address the potential relative influences of summer monsoon *versus* winter flood events.

RESULTS

Numbers of most fish species decreased markedly between Spring 1994 and 1995 sampling; only green sunfish increased (Table 1). Native fish decreased by between 56 and 99% (mean, 83%) non-natives by 11 to 100%. Yellow bullhead was reduced only 11% and smallmouth bass 28%. Red shiner decreased by 93%, and no fathead minnows were found in Spring 1995.

Bi-annual sampling at the upper- and lower-most sites in the river offers a more detailed picture of population dynamics (Table 2). All native species, except roundtail chub, declined from Spring 1994 to Autumn 1994 at the Burnt Ranch site, decreased further by Spring 1995 before increasing in Autumn 1995 and Spring 1996. Most non-native species varied little between samples. The introduced red shiner population at Burnt Ranch held stable between Spring and Autumn 1994 sampling and then declined markedly (>80%) by Spring 1995 only to then increase markedly (95%) by Autumn 1995. By comparison, at the Sycamore site, red shiner increased over 100-fold between Spring and Autumn 1994 only to collapse (-99%) to very low population levels in 1995 and 1996. At Sycamore, common carp numbers were low from Spring 1994 to Spring 1996. Similarly, smallmouth bass numbers remained low during all sample periods, disappeared at Sycamore in Spring 1995, only to increase slightly by Autumn 1995 at this site. Green sunfish were absent at these two sites in 1994, increasing at the Burnt Ranch site by Autumn 1995. Numbers of yellow bullhead were generally low and declined steadily from Spring 1994 to

Spring 1995 before increasing in Autumn 1995. Fathead minnows were essentially absent from all sampling.

At all seven sites, native fishes predominated (82 to 85%) in numbers over the non-native species (15 to 18%) (Table 1; Fig. 2). However, such predominance of native species was variable, comprising 54% of the fish community in Autumn 1994 and increasing to 85% over the entire study area by Spring 1995. By Autumn 1995 the native fish community had increased to comprise 88% of the total fish community. Fish population data collected in an independent study of the Verde River in 1986 (USFWS 1989) indicated that native species comprised 88% of the fish community (Table 3). Sampling by the Arizona Game and Fish Department immediately after the 50+year flood event in Spring 1993 (Fig. 3) indicated natives were the dominant component (96%) of the fish community by Autumn 1993 (Table 3).

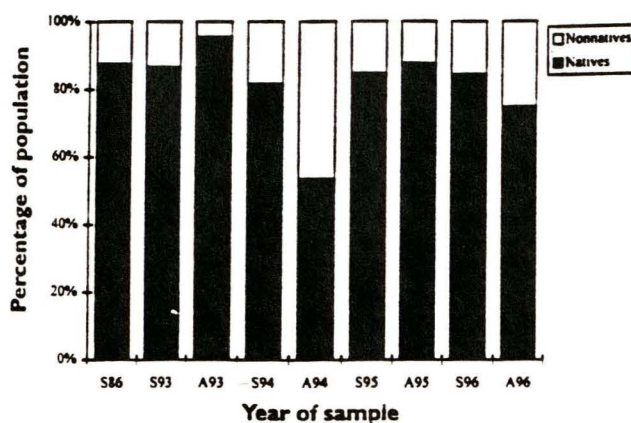


Fig. 2. Proportional numbers of native to non-native species of fishes in the upper Verde River, 1986–1996. Ratios for 1993 through 1995 are for Spring (S) and autumn (A), respectively.

Table 2. Fish community dynamics based on Spring 1994, Autumn 1994, Spring 1995, Autumn 1995, and Spring 1996 sampling at Burnt Ranch (BR) and Sycamore Creek (SC) study sites. The total number at the two sites is indicated. Order of species as in Table 1

Species	Spring 94			Autumn 94			Spring 95		
	BR	SC	Total	BR	SC	Total	BR	SC	Total
Native species									
CACL	339	379	718	31	93	124	15	29	44
CAIN	278	223	551	214	25	239	60	37	97
GIRO	15	165	180	50	17	67	3	104	107
AGCH	1072	1	1073	94	0	94	0	0	0
MEFU	257	92	349	93	0	93	33	17	50
RHOS	0	19	19	0	1	1	0	0	0
Introduced species									
CYLU	39	3	42	50	395	445	7	5	12
CYCA	1	4	5	67	1	68	0	2	2
MIDO	2	3	5	2	3	5	3	0	3
LECY	0	0	0	0	0	0	0	2	2
AMNA	2	10	12	1	6	7	2	1	3
PIPR	0	0	0	5	0	5	0	0	0

Species	Autumn 95			Spring 96		
	BR	SC	Total	BC	SC	Total
Native species						
CACL	44	77	121	79	38	117
CAIN	103	93	196	92	41	133
GIRO	40	6	46	23	25	48
AGCH	397	0	397	91	1	92
MEFU	290	0	290	33	51	84
RHOS	0	12	12	0	0	0
Introduced species						
CYLU	151	7	158	88	9	97
CYCA	3	0	3	1	1	2
MIDO	0	9	9	5	0	5
LECY	7	0	7	1	0	1
AMNA	5	8	13	1	0	1
PIPR	0	0	0	0	0	0

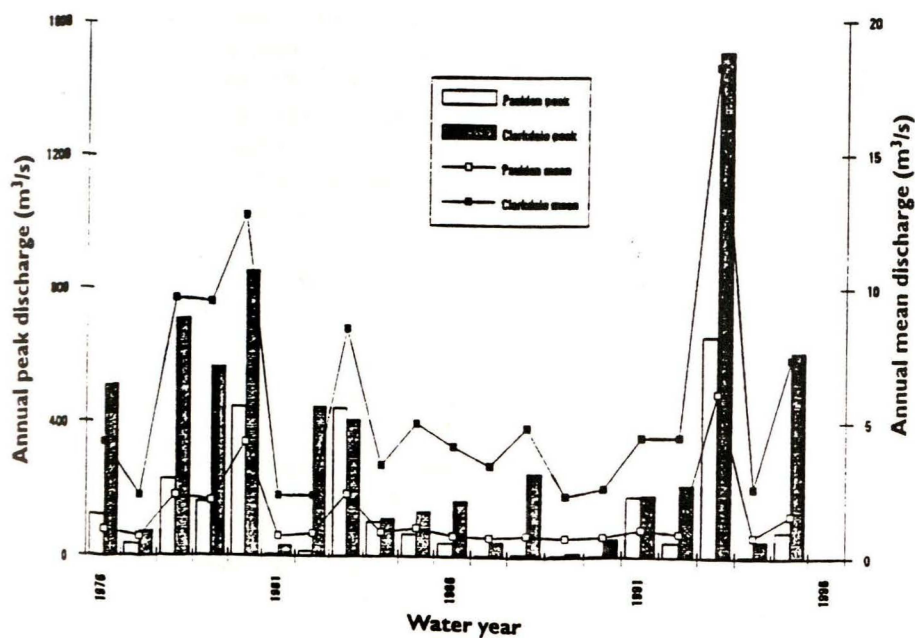


Fig. 3. Annual peak and mean annual discharge at Paulden and Clarkdale gauges during water years 1976 to 1995 (US Geological Survey records).

Table 3. Relative percentages of the fish community collected in the Verde River 1986–96. Species designation as in previous tables except for ICPU, channel catfish; species ratios (%) are indicated by season

Species	Spr 86	Spr 93	Aut 93	Spr 94	Aut 94	Spr 95	Aut 95	Spr 96
Native species								
CACL	18.7	26.2	20.9	30.4	10.8	25.7	14.7	21.3
CAIN	13.0	17.9	27.8	20.8	20.8	25.3	16.8	29.6
GIRO	11.0	1.8	1.6	8.9	5.8	26.8	4.3	11.7
AGCH	25.1	25.5	34.4	15.2	8.2	0.9	26.6	12.7
MEFU	19.1	3.7	2.5	4.9	8.1	5.7	20.1	6.3
RHOS	0.8	10.0	8.6	2.0	0.1	2.0	4.7	3.0
%	88.0	85.0	96.0	82.0	54.0	85.0	88.0	85.0
Non-native Species								
CYLU	6.6	7.4	2.9	16.9	38.8	7.6	10.6	12.4
CYCA	0.1	1.3	0.4	0.3	5.9	0.5	0.2	0.6
MIDO	0.6	1.5	0.0	0.0	0.0	2.3	0.5	0.3
AMNA	3.3	1.1	0.6	0.4	0.6	2.3	1.0	0.3
ICPU	0.4	0.4	0.0	0.1	0.0	0.2	0.0	0.0
PIPR	0.0	3.1	0.1	0.1	0.4	0.0	0.0	0.0
%	12.0	15.0	4.0	18.0	46.0	15.0	12.0	15.0

DISCUSSION

Minckley and Meffe (1987) contrasted differences in flood hydrology between southwestern arid lands and those of lowland mesic regions of the central and eastern USA. They reported that most of the annual water yield in southwestern systems was produced during high discharges in brief periods of time, whereas low discharges produced a far greater proportion of total yield from mesic systems. They suggested the differential effects of these patterns on fishes was significant to the native: non-native mix of species now present in the Southwest. Minckley and Meffe (1987) concluded that native southwestern fishes were better adapted to withstand the effects of large floods than non-native fishes. During severe floods, non-native species were either displaced or killed, whereas native species maintained position in or adjacent to channel habitats, persisted in micro-refuges, or rapidly recolonized if displaced. Based on our data from the Verde River following 50+ and 7-year flood events, it appears that floods of this magnitude negatively impact and dramatically reduce both native and non-native fish communities. The 7-year event in Winter (March) 1995 reduced the native fishes in the upper Verde by 86% and the non-native community by 89%. However, the native: non-native community composition ratio remained almost identical between years, respectively (82:18 and 85:15). Further, by Spring 1996 the native fish component had increased, on average, over 700%; non-natives, despite marked increases in red shiner and smallmouth bass, increased by an average of only 71%.

Data from work previous to our study indicated that following almost a decade (1984–92) of drought and low flows, non-natives only comprised 12% of the total fish community (Marty Jackle, US Bureau of Reclamation, pers. comm.; USFWS 1989). Following the massive flooding in 1993, non-natives were reduced to only 4% of the total fish community. Therefore, Minckley and Meffe's hypotheses on the mechanism of

interactions and control of the native and non-native fish community in the Southwest may be partially correct. First, not only the non-natives are impacted by flooding, but the natives as well. Immediately post-flood, native fish rebound rapidly; non-natives also increase but more slowly, and some ratio of the two is established. The ratio appears, in part, to be dependent on the subsequent annual hydrographs, but has been consistently 85:15, native to non-native species. However, the fact that non-natives comprised only 12% of the total fish community in the late 1980s (see above) after a four-year period of low flow suggests other mechanisms may influence the native: non-native ratio in the Verde River.

Green sunfish was the only species to increase in absolute numbers in the Verde River fish community between 1994 and 1995 (5–29). We suggest it was a result of a differential magnitude of flooding over the entire study area. The flood in 1995, in contrast to that in 1993, affected mostly the reach of river downstream from Sycamore creek (i.e. the Clarkdale gauge). The majority (26–29 individuals) of the increase in green sunfish numbers occurred in the upper reaches of the river (sites 2–5). The peak discharge at these sites, as indicated by the Paulden gauge, suggests that flows were either below the threshold to negatively impact this species, or of the proper magnitude to overflow ponds on the watershed and introduce sunfish into the river, or a combination of both. Similarly, we suggest that the input of fathead minnow from watershed stock tanks may have been the reason for the appearance of this species immediately post-flood in the Spring 1993 sample at Burnt Ranch. By Autumn 1993, fatheads had disappeared and the native fish component reached 96% of the fish community.

Precipitation and stream hydrographs are stochastic and unpredictable in the Southwest and floods of significance (i.e. $>400 \text{ m}^3 \text{ sec}^{-1}$) in the upper Verde River appear to occur randomly. Cycles of flood and drought, however, are evident,

and ensuing years of low flow are now probable following the 1993 and 1995 floods. Accordingly, based on the historic hydrograph, the probability for a period of low flows from now to the year 2000 increases. Consequently, an excellent opportunity exists to study in detail native *versus* non-native fish interactions during reduced flooding and low flow conditions.

Typical of desert streams and rivers, flooding can be significant in the Verde River. Larger floods, estimated to have recurrence intervals of 50–60 years (US Geological Survey 1992), appear to 'reset' the biological and geomorphological reference baseline. That is, riparian vegetational succession was set back to a base level, channel morphology was modified, and stream-bed materials were sorted and rearranged. Flood events in 1993 (50+ year) and 1995 (7-year) altered channel morphology and aquatic habitats by eroding stream banks and restructuring and 'invigorating' substrate materials.

After each flood, total abundance of fish was reduced and the population structure of the various species altered. Based on data collected in Winter 1993 and 1995 flood events, both 50+ and 7-year return intervals reduced both native and non-native species. Nevertheless, natives still comprised 85% or more of the fish community. Following these events the natives rebound quickly in numbers in response to the restructuring or 'invigorating' of substrate materials (Mueller 1984). Spawning success and recruitment probably contribute to the quick re-establishment of native fish populations. Although non-native species begin to increase at the same time (Table 2,) their lowered numbers also would favour a rapid increase in native fish numbers because of the lack of competition and more probably predation (Minckley 1983; Rinne and Alexander 1995; Rinne 1995b). Further, recovery rates of non-native fish populations were variable. Red shiner were markedly reduced by the 7-year flood event, common carp were less reduced, and smallmouth bass sustained their numbers.

Long-term (>5 years) studies of native fish populations in low, desert rivers of similar or larger-size are non-existent. Several long-term studies in smaller streams are ongoing: Aravaipa Creek has a 25-year record of fish community dynamics (W L Minckley pers. comm.), and there are 10-year records of native fishes and habitat associations for several streams in the upper San Francisco and Gila River drainages in New Mexico (D L Propst pers. comm.). Whereas these existing data sets may be useful for comparing effects of floods and droughts on native species, none of the studies are in stream reaches that have a significant non-native fish component. Thus the Verde River project is unique in the Southwest and offers an excellent opportunity to test and refine current hypotheses relative to native:non-native fish interactions as influenced by drought and flooding over the long term.

REFERENCES

- Miller, R R (1960). Man and the changing fish fauna of the American southwest. *Papers of the Michigan Academy of Science, Arts, and Letters* 46. 365–404.
- Minckley, W L (1983). Status of the razorback sucker, *Xyrauchen texanus* (Abbott), in the lower Colorado River basin. *The Southwestern Naturalist* 28. 165–87.
- Minckley, W L and Alger, N T (1968). Fish remains from an archaeological site along the Verde River Yavapai County, Arizona. *Plateau* 40. 91–7.
- Minckley, W L and Deacon, J E (1968). Southwestern fishes and the enigma of 'Endangered species'. *Science* 159. 1424–32.
- Minckley, W L and Douglas, M E (1991). Discovery and extinction of western fishes: a blink of the eye in geologic time. In 'Battle Against Extinction'. (Eds W L Minckley and J E Deacon) pp. 7–17. (University of Arizona Press: Tucson).
- Minckley, W L and Meffe, G K (1987). Differential selection by flooding in stream fish communities of the arid American southwest. In 'Community and Evolutionary Ecology of North American Stream Fishes'. (Eds W J Matthews and D C Heins) pp. 93–104 (University of Oklahoma Press: Norman).
- Mueller, G (1984). Spawning by *Rhinichthys osculus* (Cyprinidae) in the San Francisco River, New Mexico. *The Southwestern Naturalist* 29. 354–6.
- Rinne, J N (1991). An approach to management and conservation of a declining regional fish fauna: southwestern USA. *Proceedings Symposium on Wildlife Conservation, 5th International Conference Zoology* 5. 56–60.
- Rinne, J N (1993). Declining southwestern Aquatic habitats and fishes: Are they sustainable? Sustainability symposium. USDA Forest Service General Technical Report RM-247. 256–65.
- Rinne, J N (1995a). The effects of introduced fishes on native fishes: Arizona, southwestern United States. In 'Proceedings of the World Fisheries Congress, Theme 3'. (Ed D P Phillips) pp. 149–59 (Oxford and IBH Publishing Company: New Delhi).
- Rinne, J N (1995b). Interactions of predation and hydrology on native southwestern fishes: Little Colorado spindace in Nutrioso Creek, Arizona. In 'Hydrology and Water Resources in Arizona and the Southwest'. Vols 22–25. pp. 33–38. Proceedings of the 1995 meeting of the Arizona Section, American Water Resource Association and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 22, 1995, Flagstaff, Arizona. 114 pp.
- Rinne, J N. and Alexander, M A (1995). Non-native salmonid predation on two threatened native species: Preliminary results of field and laboratory studies. *Proceedings Desert Fishes Council* 26. 114–16.
- Rinne, J N. and Minckley, W L (1991). Native fishes of arid lands: A dwindling resource of the desert southwest. USDA Forest Service General Technical Report RM-206. 1–45.
- Rinne, J N. and Stefferud, J A (1996). Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona. In 'Hydrology and Water Resources in Arizona and the Southwest'. Vol. 26. pp. 13–22 (American Water Resources Association and the Hydrology Section Arizona-Nevada Academy of Sciences: Flagstaff).
- Stefferud, J A. and Rinne, J N (1995). Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes. In 'Hydrology and Water Resources in Arizona and the Southwest'. Vols 26–32. pp. 22–25. (American Water Resources Association and the Hydrology Section, Arizona-Nevada Academy of Sciences: Flagstaff).
- US Fish and Wildlife Service (1989). Fish and wildlife coordination act substantiating report: Central Arizona Project, Verde and East Verde River water diversions, Yavapai and Gila counties, Arizona.
- US Geological Survey (1992). Basin characteristics and streamflow statistics in Arizona as of 1989. US Geological Survey, Water-Resources Investigations Report 91–4041.

RELATIONSHIPS OF NATIVE FISHES AND AQUATIC MACROHABITATS IN THE VERDE RIVER, ARIZONA

John N. Rinne¹ and Jerome A. Stefferud²

There is limited information on the relative, quantitative and physical habitat utilization of native fishes within large desert river systems in the Southwest. Rinne (1992) delimited habitat for a native fish community in a small desert stream, Aravaipa Creek, and presented limited information on two native species in larger southwestern rivers (Rinne 1989, 1991). However, studies of the fish fauna in a large river are currently in progress on the upper Verde River, Yavapai County, Arizona (Stefferud and Rinne 1995). These studies, in part, are designed to define habitat affinities of native fishes in this reach of yet undammed, free-flowing river. Six native species remain in this reach, plus an equal number of non-native, introduced species (Stefferud and Rinne 1995). All appear to occupy habitats defined largely by velocity, and to a lesser extent, depth of water. Because of the paucity of information, velocity and depths of five qualitatively defined aquatic macrohabitats are being collected in the upper Verde River. In part, this ongoing study is designed to define aquatic macrohabitats in this reach of river and to delimit fish utilization of these habitats.

This paper has three objectives. The physical characteristics of these aquatic macrohabitats are defined, the associations with and utilization of these habitats by the respective species are delimited, and the significance of these data and their management implications are evaluated.

Study Area

An introduction to and a map of the general study area can be found in Stefferud and Rinne (1995). In brief, the upper Verde River is one of few remaining undammed rivers in the Southwest. Base flows range from 0.57 m³/sec at the Paulden stream gage in the upper portions of the study areas to 1.42–1.70 m³/sec at the Sycamore Creek site (Stef-

ferud and Rinne 1995; Rinne and Stefferud in press). The study of this stream reach is programmed for 10 years.

Methods

Data have been collected at seven established sampling sites since spring 1994, following a 50± year flood event in winter of 1992–93. The data presented represent the first two years (1994–95) of a 10-year programmed effort. Study reaches are sampled twice a year, in spring (April) and autumn (October). All seven sites are sampled in the spring; however, to reduce sampling stress on fish populations, only the uppermost and lowermost sites are sampled during autumn. Aquatic macrohabitats are qualitatively defined and physical measurements are normally taken in spring only. However, in the event of summer flooding, which may alter physical habitat, physical measurements would also be taken in autumn. We have visually (qualitatively) designated four major lotic or flowing water habitats: high-gradient riffles, low-gradient riffles, runs, and glides. Almost a dozen different pool habitats are being used in describing lentic or still-water macrohabitats in this paper; we classify all into one category—pool. Length of each habitat is measured and three equidistantly spaced transects recording width, depth, and velocity are measured each spring. Initially, only general categorization of stream substrates was made.

In spring 1996, we commenced to move from qualitatively to quantitatively defining macrohabitats in the Verde River. We sampled 14 microhabitats (see Neary et al. this volume) at the Burnt Ranch site in order to more validly and unequivocally define macrohabitats (see Figures 1 and 2). Accordingly, laser survey technology was employed to more accurately and quantitatively define gradients of the respective habitat types. In addition, pebble counts (Bevenger and King 1995) are being used to better define substrate composition of each habitat. Four major substrate cat-

¹USDA Forest Service, Rocky Mountain Forest & Range Experiment Station, Southwest Forest Science Complex, Flagstaff, AZ.

²USDA Forest Service, Tonto National Forest, Phoenix, AZ.

gories are employed in analyzing substrate: sand (≤ 2 mm), gravel (3–64 mm), pebble (65–126 mm) and cobble/boulder (>126 mm). Fishes in each habitat were captured using a combination of backpack DC electrofishing gear and seines (3–13 mm mesh). Only fishes captured in 1994–95 sampling were used to define macrohabitat use by respective species (see Figures 3–5).

Results

Based on visual (qualitative) designation, pools and glides had the greatest mean depth and the least mean velocity (Table 1). The range of velocity and depth of the four lotic macrohabitats varied widely; glides were greatest in mean and median depth, and low-gradient riffles were least (LGR). No trend was obvious for depth of these habitats. Mean velocity of lotic habitats increased from glides through high-gradient riffles (HGR).

By comparison, laser-estimated gradients at the 14 microhabitat sites parallel mean velocities of the habitat types (Figure 1). A working hypothesis for categorizing macrohabitats based on percent gradient is glides = $\leq 0.3\%$, runs = $0.3\text{--}0.5\%$, LGR = $0.6\text{--}1\%$, and HGR $>1\%$. Mean velocities also fall within these same categories. Substrate composition estimated within the 14 microhabitat sites is

directly legislated by velocity of water (Hynes 1972). Substrate also demonstrated patterns relative to the major habitat types in the Verde. That is, sand and gravel decreased from glides through HGR habitats and pebble and cobble/boulder substrate increased in the same sequence (Figure 2).

Table 1. Mean velocity and depth characteristics of aquatic macrohabitat types in the upper Verde River, 1994. Number of macrohabitats sampled is in brackets, and medians are in parentheses.

Habitat type	Depth (cm)	Velocity (cm/sec)
Pool [39]	46 (40) 20–90	22 (2) 0–70
Glide [60]	55 (50) 25–43	22 (25) 15–26
Run [94]	42 (37) 20–80	48 (43) 20–80
Low-Gradient Riffle [96]	25 (25) 10–41	59 (51) 30–80
High-Gradient Riffle [98]	34 (30) 20–60	87 (90) 76–150

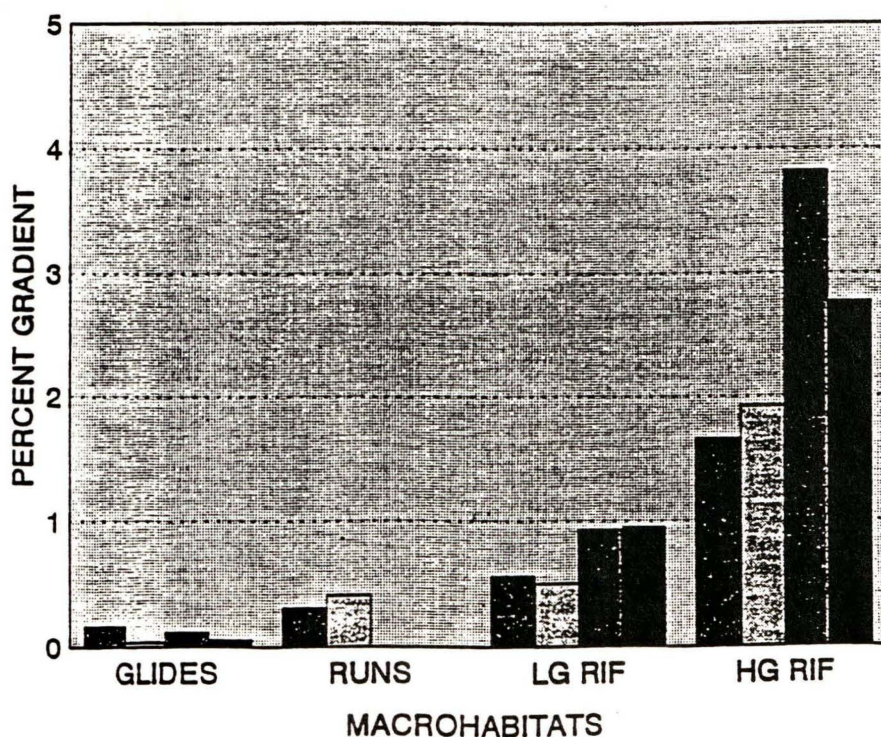


Figure 1. Comparison of laser-generated gradients at the 14 microhabitats at Burrnt Ranch, upper Verde River, spring, 1996.

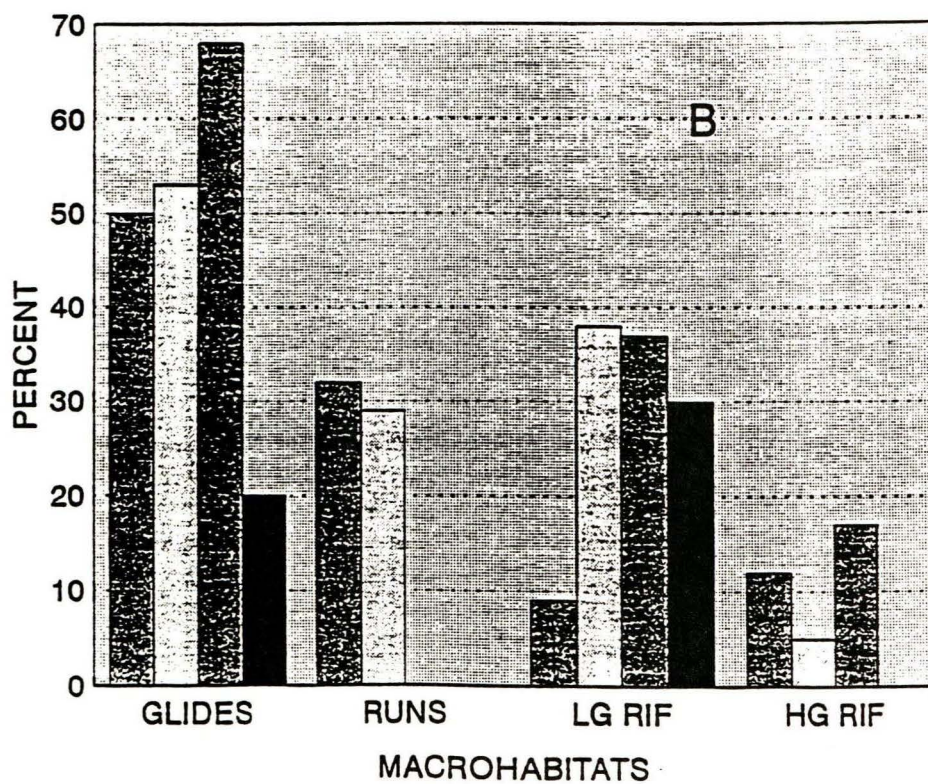
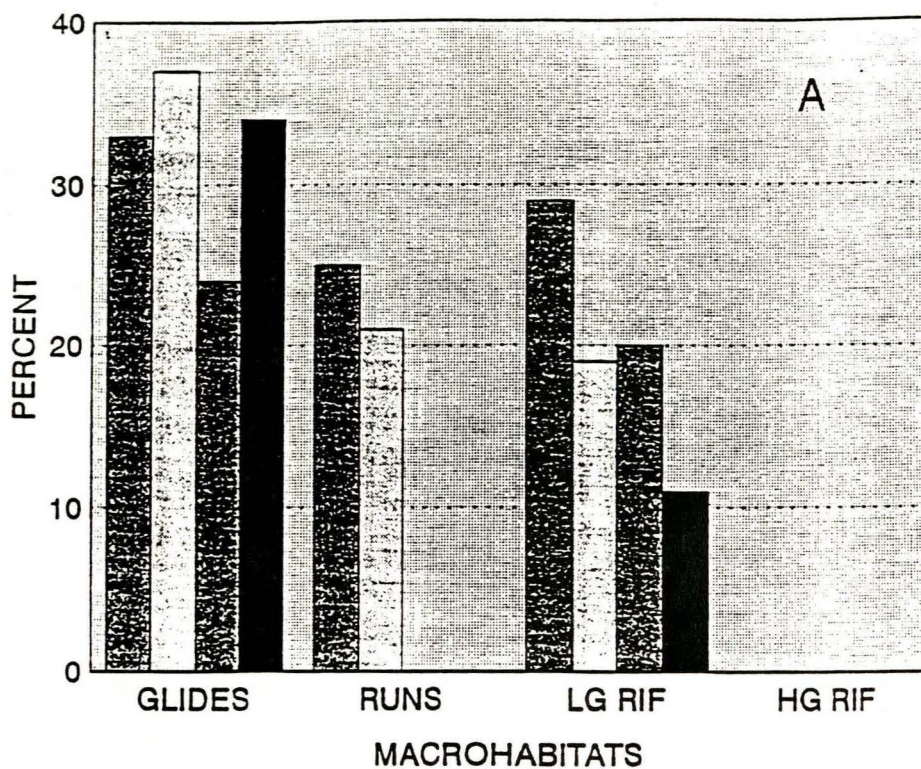


Figure 2. Comparison of (A) sand, (B) gravel, (C) pebble, and (D) cobble/boulder components in 14 aquatic microhabitats in the upper Verde River, spring 1996.

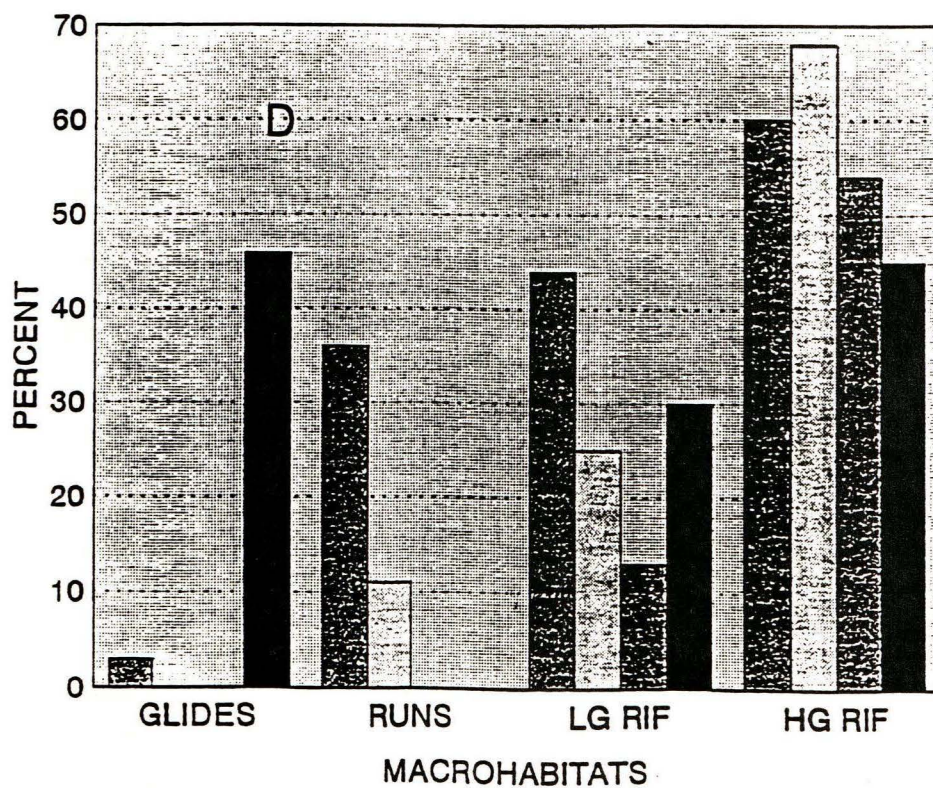
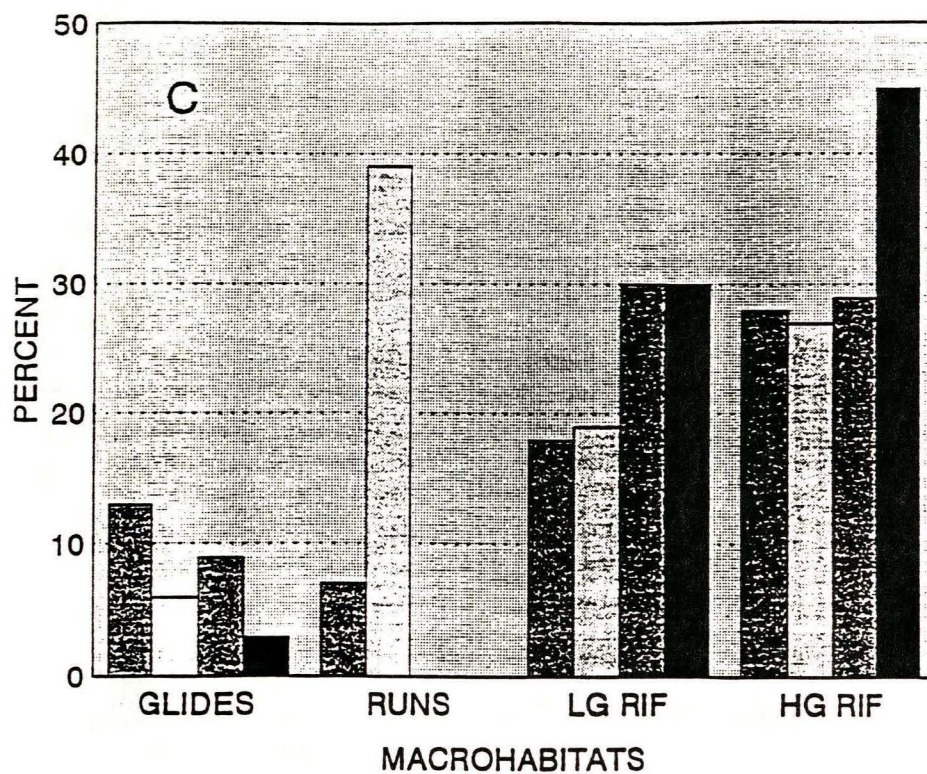


Figure 2 (continued)

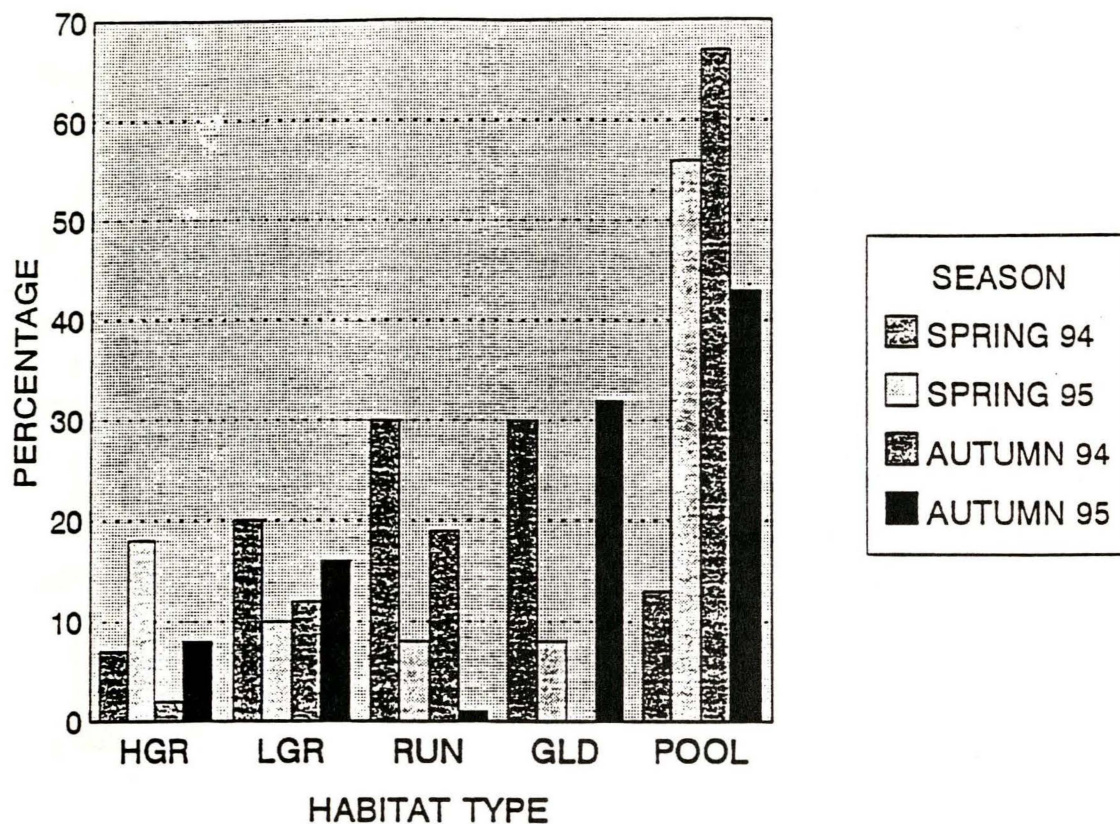


Figure 3. Comparison of macrohabitat utilization by the native fish community in the upper Verde River, 1994-95.

Information collected on fish numbers at the seven sites over four sample periods gives the appearance that, with the exception of spring 1994 sampling, the native fish community is a pool-dwelling assemblage (Figure 3). However, this is attributable to occupation of pool type habitats by two of the three larger sized species, Sonora sucker and roundtail chub (*Catostomus insignis* and *Gila robusta*, respectively), which comprised almost half of the total number of fish collected during the four sampling periods (Figure 4). By comparison, over half (56%) of desert suckers utilized HGR and LGR macrohabitats. Similar to desert sucker, most (90%) speckled dace also were collected in HGR and LGR habitats (Figure 5). Longfin dace predominantly occupied LGR and run habitat. Finally, most (80%) of the federal and state-threatened spinedace were collected in run and glide habitat (Figure 5).

Discussion and Conclusions

Qualitatively defined aquatic macrohabitats appear to be reasonably valid based on velocity, substrate, and gradient criteria at the 14 micro-

habitat sites. However, more data are needed. For example, physical similarity of pools and glides suggests that refinement is needed for these two habitat types. Also, based on macrohabitat data collected in 1994-95, spinedace preferred runs and glide habitats. Microhabitat sites suggested that spinedace prefer runs and low-gradient riffles (Neary et al., this volume).

A working categorization for gradients and velocities is proposed; however, more quantitative microhabitat data are needed to validly and unequivocally define the respective macrohabitats. Depth appears to be much less of a definitive physical characteristic than velocity or gradient for macrohabitat designation. We acknowledge that only limited microhabitat data (spring 1996) are available on substrate characteristics. However, substrate is certainly delimited by stream gradient and velocity of water; this physical feature may be very important both in habitat definition and in legislating fish distribution.

Gradient may be especially important in defining habitats for the habitat specialists such as spinedace. The importance of substrate in spike-

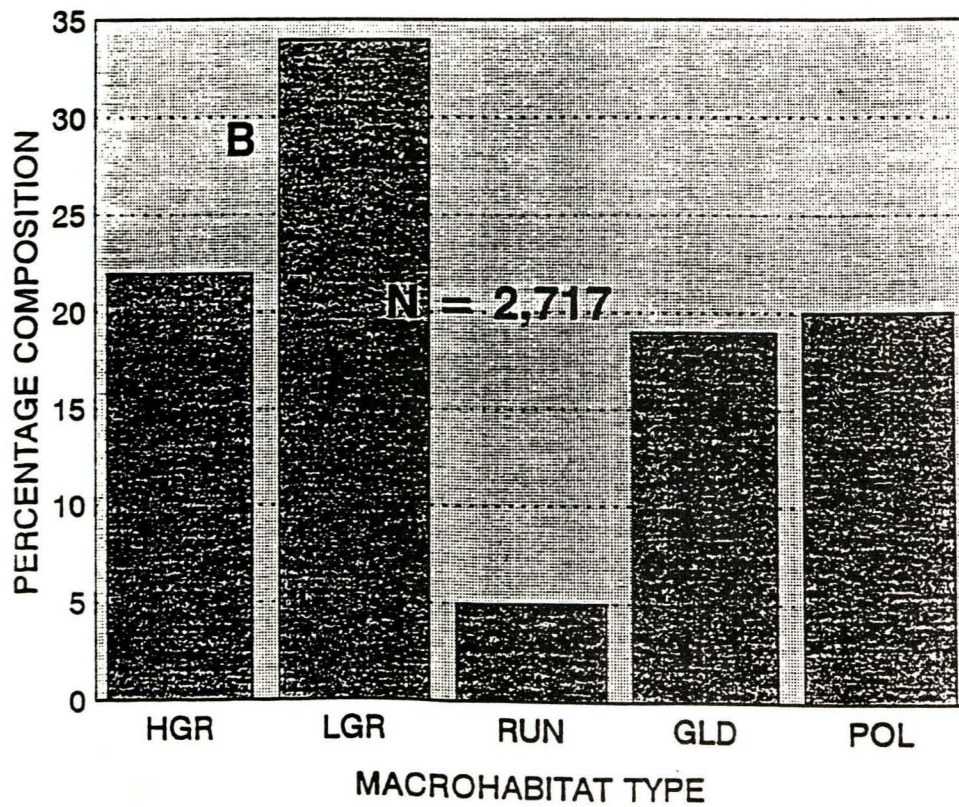
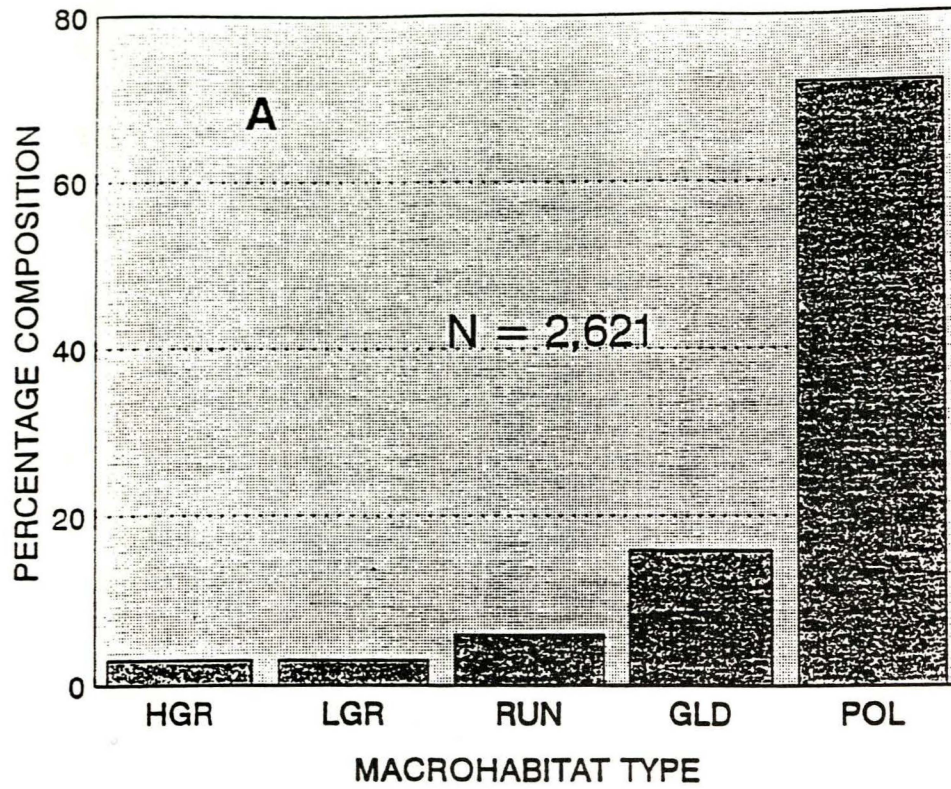


Figure 4. Macrohabitat utilization of the (A) Sonora and (B) desert suckers and (C) roundtail chub in the upper Verde River, 1994–95.

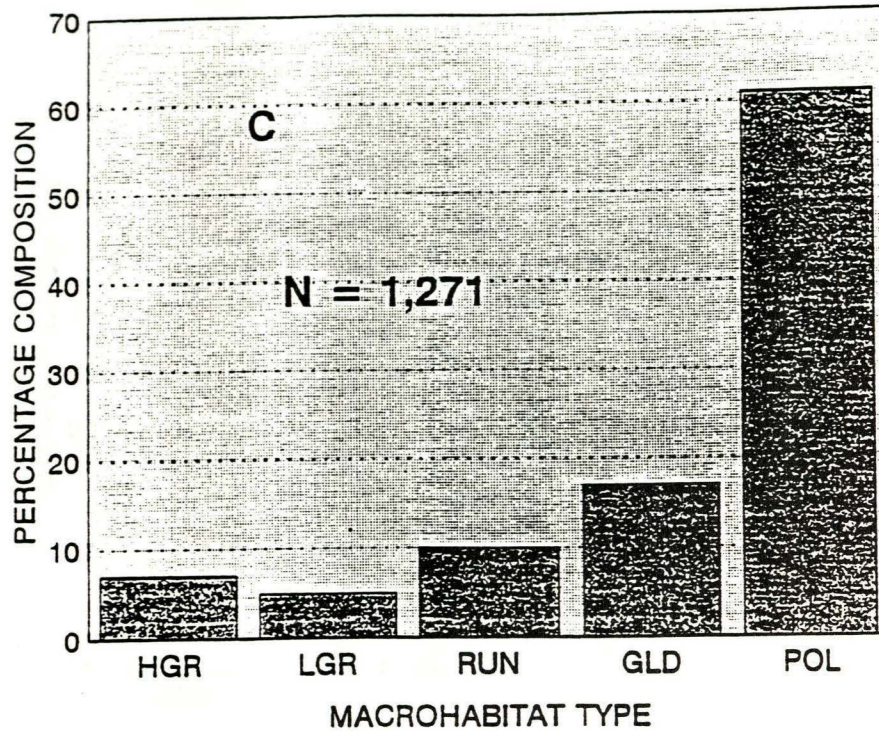


Figure 4 (concluded).

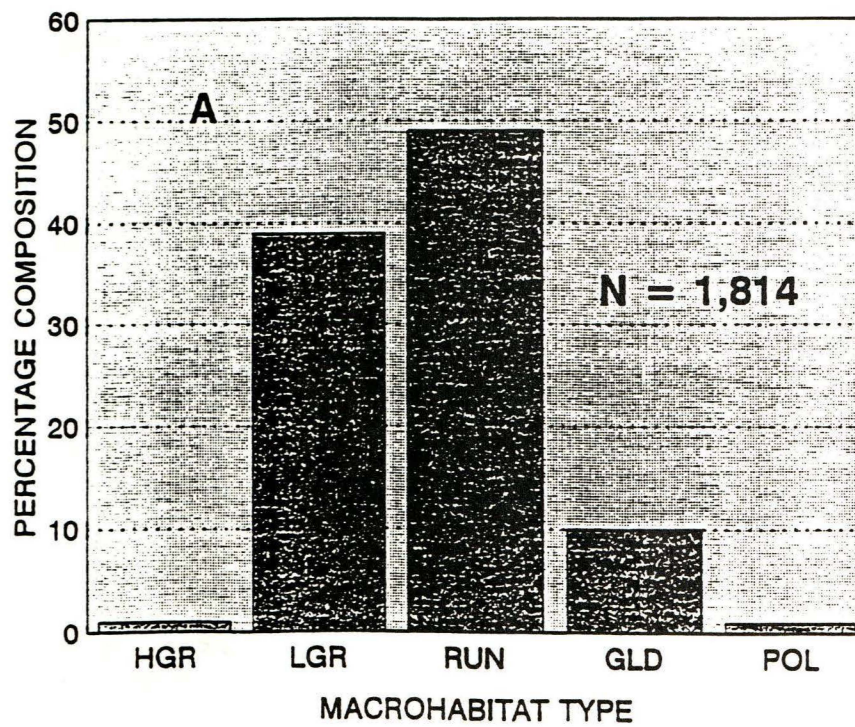


Figure 5. Macrohabitat utilization by the (A) longfin dace, (B) spinedace, and (C) speckled dace in the upper Verde River, 1994-95.

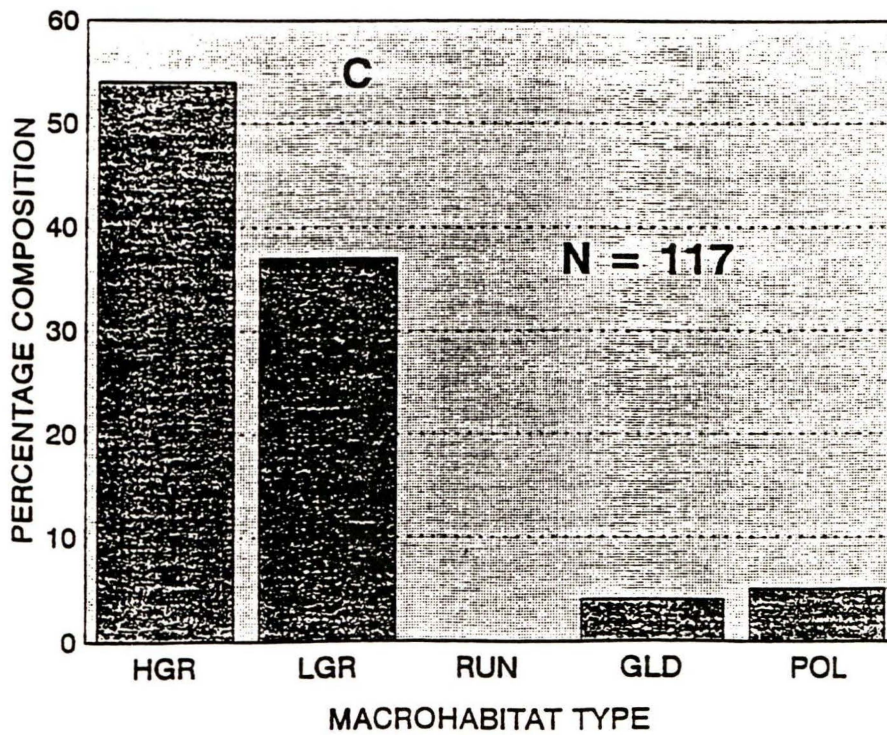
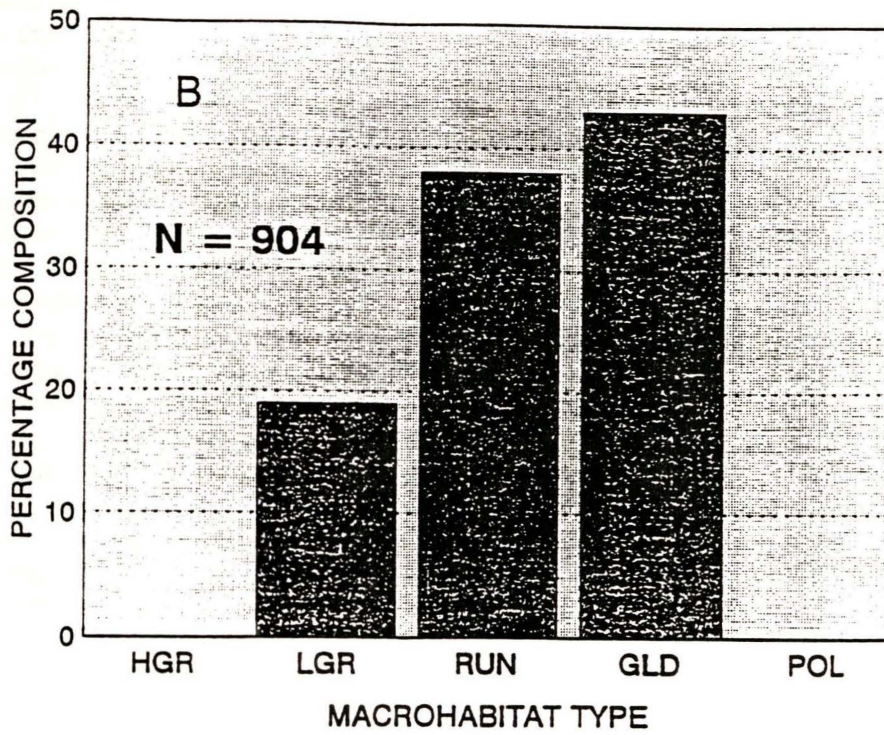


Figure 5 (concluded)

dace distribution and abundance is beginning to be documented for this species in the Verde River (Neary et al., this volume) and substantiates earlier work by Barber et al. (1966) and Barber and Minckley (1967). This threatened species occupies gravel to pebble substrates (Rinne 1991, 1992; Neary et al., this volume). Both our spring 1996 data in the upper Verde River and that collected in April 1985 (Rinne 1991) indicate that gravel/pebble (3–64 mm) substrates are particularly important for spawning habitat of spikedace.

Our data on the upper Verde River suggest that gravel/pebble substrate habitats are concentrated in runs and low-gradient riffle habitats (Figure 2). In part, gradient and the resulting increase in velocity are primary influencing factors, but channel narrowing also increases velocity. That is, if the same volume of water passes through a narrower and deeper channel, velocity is increased and the deposition of gravel/pebble substrate is enhanced, whereas the deposition of sand substrate decreases. The process of channel narrowing requires streambank stability. Herbaceous native aquatic vegetation has been suggested as a very important component contributing to streambank stability (Medina 1995). Land-use practices that remove vegetation can reduce streambank stability and the channel-narrowing capacity of a stream (Neary and Medina 1995). Studies examining the linkages of grazing, streambank stability and morphology, vegetation composition and density, fish habitat, and fish populations on the upper Verde are being initiated and will conceivably provide answers on the linkage between grazing and spikedace habitat and populations. Land managers conceivably can use this information to implement grazing strategies in time and space that sustain spikedace habitat.

Our data on the upper Verde (Figure 5) and that of other authors (Barber and Minckley 1967; Minckley 1973) document the close relationship of longfin dace with low-velocity, sandy bottom habitats. The occupation of pool-type habitats by almost 60 percent of Sonora sucker and roundtail chub also suggests the importance of the lentic component of the upper Verde ecosystem. The other three species are lotic water inhabitants. In summary, the native fish community in the Verde River basically utilized the same relative macrohabitats as recorded for the same community in Aravaipa Creek (Rinne 1992). However, more data are needed on the relative proportions of the major

aquatic macrohabitat types in the upper Verde river before irrefutable statements can be made on habitat use by the respective species. That is, at present, we are assuming that the habitats we have sampled at the seven sites are representative of those available in the 60-km reach of the upper Verde.

Flood events may drastically alter extant macrohabitats, resulting in a new assemblage of habitats within established study reaches. The more drastic the changes in the habitats within these reaches, the greater the changes in the relative composition of the fish community. Such events and resultant changes provide the opportunity to validate both micro and macrohabitat vs. fish relationships. Flood events such as those in winter 1993 appear essential for creating and maintaining pools in addition to inducing increased or successful spawning and recruitment (Stefferud and Rinne 1995; Rinne and Stefferud, in press).

There is a need for additional refinement in the definition of the physical characteristics of the aquatic macrohabitats in the upper Verde; however, gradient appears to be of great importance in fish distribution. Gradient can be very precisely defined by laser technology. Further, gradient directly influences the velocity of water and substrate composition. Therefore, we suggest that this physical feature may be the controlling factor for quantitatively delimiting macrohabitats in this and perhaps other desert streams and rivers. Nevertheless, additional quantitative, objective data on macrohabitat utilization by the respective members of the native and non-native fish community will be needed to compare with and validate data collected to date.

References Cited

- Barber, W.E. and W.L. Minckley. 1966. Fishes of Aravaipa Creek, Graham and Pinal counties, Arizona. *The Southwestern Naturalist* 11(3):313–324.
- Barber, W.E., D.C. Williams and W.L. Minckley. 1966. Biology of the Gila spikedace, *Meda fulgida*, in Arizona. *Copeia* 1970:9–18.
- Bevenger, G.S. and R.M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. USDA Forest Service Research Paper RM-319:1–17.
- Hynes, H.B.N. 1972. The ecology of running waters. University of Toronto Press. 555 pp.
- Medina, A.L. 1995. Native aquatic plants and ecological condition. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report 272:329–335.

- Neary, A., J.N. Rinne, J.A. Stefferud and D.G. Neary. In press (this volume). Physical habitat use by spike-dace in the upper Verde River, Arizona. In *Hydrology and Water Resources in Arizona and the Southwest*, Vol. 26. Proceedings of the 1996 meetings of the Hydrology Section, Arizona-Nevada Academy of Sciences, April 20, 1996, Tucson, Arizona.
- Neary, D.G. and A.L. Medina. 1995. Geomorphic response of a montane riparian habitat to interactions of ungulates, vegetation, and hydrology. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report 272: 143-147.
- Rinne, J.N. 1989. Physical habitat use by loach minnow, *Tiaroga cobitis* (Pisces: Cyprinidae), in southwestern desert streams. *The Southwestern Naturalist* 34(1): 109-117.
- Rinne, J.N. 1991. Physical habitat use by spikedace, *Meda fulgida*, in southwestern desert streams with probable habitat competition by red shiner, *Notropis lutrensis* (Pisces: Cyprinidae). *The Southwestern Naturalist* 36(1): 7-13.
- Rinne, J.N. 1992. Physical habitat utilization of fish in a Sonoran Desert stream, Arizona, southwestern United States. *Ecol. Freshwater Fishes* 1: 1-8.
- Rinne, J.N. and J.A. Stefferud. In press. Factors contributing to collapse yet maintenance of a native fish community in the desert Southwest (USA). Second World Fish Congress, Brisbane, Australia. July 28-August 2, 1996.
- Stefferud J.A. and J.N. Rinne. 1995. Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes. Pages 26-32 in *Hydrology and Water Resources in Arizona and the Southwest*, Vols. 22-25. Proceedings of the 1995 meetings of the Arizona Section, American Water Resource Association and the Hydrology Section, Arizona-Nevada Academy of Science, April 22, 1995, Flagstaff, AZ.

**WATER RESOURCES EDUCATION, TRAINING, AND PRACTICE:
OPPORTUNITIES FOR THE NEXT CENTURY**

JUNE 29-JULY 3

AMERICAN WATER RESOURCES ASSOCIATION

1997

STREAM CHANNEL AND FISH RELATIONSHIPS:

PRELIMINARY OBSERVATIONS, VERDE RIVER, ARIZONA

John N. Rinne and Daniel G. Neary¹

ABSTRACT: The Verde River is a large, low elevation desert river in west-central Arizona. Successful management of the native Threatened and Endangered (TES) fish community in the Verde watershed hinges on determining the relationship of channel geomorphology to fish habitat requirements. Much of the fish habitat literature comes from other species in montane riparian systems and may not be applicable to lower elevation streams in the arid Southwest. We determined channel type, bedload substrate composition, and fish populations in two distinct categories of stream reaches in the Verde River, Arizona. The two categories were E-type channels with native aquatic vegetation (e.g. sedges, rushes) and C-type channels without this vegetation component. Preliminary data indicate that 6 to >100 times the numbers and 4 to 50 times the biomass of native TES fishes to non-native fishes occur in C-type, non-sedge channels. By comparison, non-native fishes were equal or greater in numbers and 3 to 4 times the biomass of native species in E-type channels with sedge bank habitats. The significance of these data relative to 1) native/introduced fish interactions in the Verde, 2) riparian-stream management in this low desert river, and 3) future research needs, sampling approach, and management implications are discussed.

KEY TERMS: Southwest, desert rivers, native fishes, stream channel types

INTRODUCTION

There is limited information on the quantitative, physical habitat utilization by native fishes in low desert, riverine ecosystems in the Southwest. Rinne (1992) delimited habitat for a native fish community in a small desert stream, Aravaipa Creek, and presented information on two native species in larger southwestern rivers (Rinne, 1989, 1991).

The Verde River is a large, low elevation desert river in west-central Arizona. Successful management of the native threatened and endangered (TES) fish community in the Verde watershed depends, in part, on determining the relationships of channel geomorphology, fish habitat, and fish populations. Most of the available fish habitat literature comes from salmonid species in montane riparian systems and may not be applicable to cypriniform (minnow and sucker) species in lower elevation streams in the arid Southwest.

Studies are currently in progress on the fish fauna in the upper Verde River, Yavapai County, Arizona (Stefferdud and Rinne, 1995). These studies are, in part, designed to define habitat affinities of native fishes in this reach of free-flowing river. Six native species are extant in this reach of the Verde and two additional species are the object of re-introduction efforts (Stefferdud and Rinne, 1995). An equal number of non-native, introduced species also inhabit this reach of river (Stefferdud and Rinne, 1995). All appear to occupy habitats defined largely by velocity of water, stream substrate, and to a lesser extent, depth of water (Rinne and Stefferud, 1996; Neary et al., 1996).

¹ Fisheries Biologist and Project Leader, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, 2500 South Pine Knoll Drive, Flagstaff, Arizona 86001.

Herbaceous aquatic vegetation, largely rhizomatous graminoids (sedges, rushes, bulrushes) is increasingly being demonstrated as a vital component of proper functioning of aquatic systems (Medina, 1995). Livestock grazing is a primary, widespread land management activity on public lands in the arid West and Southwest. This land management activity may impact structure and integrity of streambanks (Neary and Medina, 1995). There is an urgent management need to know the relationship between grazing, streambank vegetation, physical habitat and fish populations in river systems of this type. Once determined, the direct effect of grazing via removal of vegetation and streambank trampling and the indirect effect of the hydrologic regime on these impacts can be established. At the present, these relationships have not been adequately demonstrated by valid, scientific studies and are mostly speculative.

Much of the fish habitat literature addressing the importance of vegetation comes from salmonid species in upper elevation, montane riparian-stream habitats from other biogeographic regions (Bjornn and Reiser, 1991). Accordingly, this information may not be applicable to lower elevation streams in the arid American Southwest. That is, overhanging, well-vegetated banks are considered essential components of streambank stability and proper functioning (Medina, 1995). In turn, well-vegetated, stable banks result in overhang that provides optimal salmonid habitat in the form of cover (Platts et al. 1983, 1987; Bjornn and Reiser, 1991). There is no information on the relationship of physical structure (e.g. overhanging banks), channel geomorphology, stream substrates and fish populations in low desert, riparian-stream ecosystems.

The linkages of vegetation, streambank structure, and fish habitat (ultimately fish populations) are not known. It is essential to determine and delineate the relationships of channel type, streambanks and fish habitat and populations as a prerequisite to any research on the linkages and effects of grazing on streambanks and ultimately on fish populations.

Preliminary data presented here are a result of research designed to determine if stable streambanks in low desert aquatic ecosystems are important in terms of fish habitat. Whether stable banks and associated instream aquatic habitat are more or equally beneficial to native or introduced species of fish is unknown. Studies are available in the Pacific Northwest and Intermountain West on the effects of grazing on streambanks and streambank vegetation (Platts, 1991). Again, applicability of these studies to the Southwest is suspect.

We determined channel type, bedload substrates, and native versus non-native, introduced fish density, biomass and diversity in two distinct categories of stream reaches in the Verde River, Arizona. The two categories were E-type channels (Figure 1; see also Rosgen, 1994) with stable streambanks occupied by native hydric vegetation and C-type channels without this vegetation component. The primary objectives of this study are to begin to: 1) define streambank morphology, instream substrates, and aquatic macrohabitats in two different channel types; and 2) define and compare fish community characteristics between the two channel types.

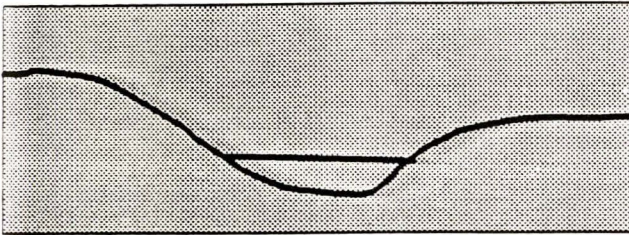
METHODS

The general area of study lies within the upper Verde River, Prescott National Forest, Yavapai County, Arizona (Stefferdud and Rinne, 1995; Figure 1). Specific sampling was conducted in June, 1996, at, and immediately below, the Rio Verde Ranch (T 18N, R 1W, SW 1/4 Section 35, Prescott National Forest Map). The site is within 1 km of the USGS Paulden streamflow gauge. Baseflow at this site is about 1.2 m³/min (20 cfs). Sampling was conducted on contiguous private and U. S. Forest Service lands within a half kilometer reach of river.

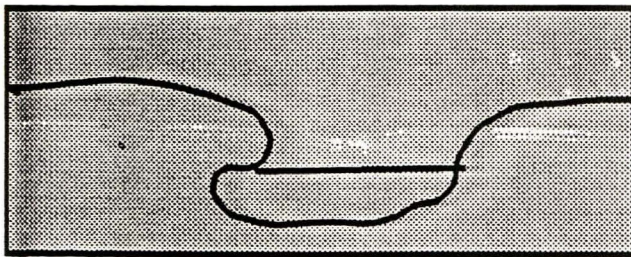
Three stream reaches were sampled within the well-vegetated, E-type, sedge lined channels; four in the non-sedge, C-type channels (Figure 1, Table 1). All C-type channels were comprised of run habitats; E-types contained one each of run, high (HGR) and low (LGR) gradient riffle habitat types (Rinne and Stefferud, 1996). Upper and lower ends of each reach were blocked with minnow seines (6.4 mm mesh). Backpack, DC electrofishing gear and bag seines (3.2 mm mesh) were used to sample fishes. All fishes in these units were captured, weighed, measured, and returned alive to the

CROSS SECTIONAL VIEW

LONGITUDINAL VIEW



STREAM CHANNEL TYPE C



STREAM CHANNEL TYPE E



Figure 1. Idealized illustrations of cross sectional and longitudinal views of E and C channel types

Table 1. Fish-channel-study section relationships, June, 1996, Verde River, Arizona.

Verde Study Sections	Native Fish		Non-Native Fish		Native/Non-Native Fish Ratios		
	Number	Mass-g	Number	Mass-g	Number	Mass-g	Species
E/I	16	92	28	347	1:2	1:4	2:3
E/II	11	108	11	357	1:1	1:3	2:4
E/III	29	1,038	7	52	4:1	20:1	3:3
E Channel Mean	19	412	15	252			
C/IV	90	2,031	10	659	9:1	3:1	4:3
C/V	291	2,168	49	346	6:1	6:1	3:3
C/VI	256	400	3	8	118:1	50:1	2:2
C/VII	328	465	30	110	11:1	11:1	4:3
C Channel Mean	241	1,270	21	281			

stream. Fish density and biomass by species and by community (native vs introduced) were calculated.

Streambank and instream morphology and bank undercut (volume of aquatic habitat) within each sample unit was estimated by standard, metric measurement with meter rules. Transects were positioned at right angles to flow. Depth and velocity of water were estimated with a meter rule and a direct reading current meter. Stream substrate was characterized by pebble count methodology (Bevenger and King, 1995). Fifty pebble count measurements were taken within each stream sample unit.

Statistical analyses for mean separations were precluded because of sample size. Means were calculated for fish biomass and numbers by channel type.

RESULTS

Fishes

In two of the three E-type, sedge bank channels, species ratios of non-native fishes was equal to or greater than native species. However, numbers of non-native species were equal to or twice the numbers of natives, and nonnatives displayed three to four times the biomass compared to the natives in the E-type channel sample units (Table 1). By contrast, in the single, HGR, E-type sample unit there were four times the number of natives compared to non-natives and 20 times the biomass (Table 1). Although species ratios were nearly equal in the C-type, non-sedge channels, native fishes predominated in both numbers (6 to 118 times) and biomass (3 to 50 times) in these sample units. Mean number and biomass of natives and non-natives was nearly equal for the E-type channels. In comparison, mean numbers and biomass of native fishes for all four C-type channels was twelve and four times, respectively, that of the non-native fish community in E-type channels.

Physical habitat

Mean widths were more variable in the E-type channels (3.4 to 6.2 m), and water velocity was greater in the LGR and HGR units. Mean velocity in the E-type channels (1.1 m/sec) was greater than in the C-type habitats (0.42 m/sec; Table 2). Velocities ranged from a low of 0.5 m/sec in section E-II to almost 2 m/sec in sample section E-III, a HGR unit. Mean undercut bank volume (1.18 m³) was markedly greater in the E-type channel reaches than in the C-type channels (0.28 m³, Table 2)

Substrate composition paralleled velocity; large-size materials (cobble and boulder) being represented in the LGR and HGR study reaches (I and III). All C-type channels contained nearly identical mean widths and water velocity and were similar in depth to the three E-type channel sample units. Similar to the single E-type channel, run habitat (Section II), all C-type, non-sedge sample units contained more than 60% combined sand-gravel substrate.

Table 2. Physical Habitat Characteristics of E- and C-type Channels, Verde River, Arizona. Substrate pebble counts (Number) are: Sand (s), Gravel (g), Pebble (p), Cobble (c) and Boulder (b).

Channel Section	Velocity m/sec	Depth m	Width m	Undercut m	s	g	p	c	b	Habitat Type
E/I	0.81	0.33	4.0	0.56	10	28	38	24	0	LGR ¹
E/II	0.53	0.31	6.2	2.10	36	64	0	0	0	Run
E/III	1.94	0.38	3.4	0.90	8	18	26	40	8	HGR ²
C/IV	0.44	0.33	4.5	0.50	22	38	8	52	0	RUN
C/V	0.41	0.25	4.0	0.52	20	58	14	6	2	RUN
C/VI	0.43	0.17	3.5	0.00	22	46	26	4	2	RUN
C/VII	0.41	0.20	3.5	0.00	16	48	36	0	0	RUN

1 low gradient riffle

2 high gradient riffle

DISCUSSION

Fish species ratios, numbers, and biomass varied among channel types, and aquatic macrohabitats. First, based on this preliminary sampling, it appears that in the C-type, non-sedge sample reaches, native species are always greater in numbers and biomass despite having similar species proportions to non-natives. All sample reaches fell within the run type aquatic macro-habitat in the C-type channels. Nevertheless, non-natives comprised twice the species and three times the biomass compared to the natives in the single E-type channel, run habitat. The HGR type habitat is typically inhabited by desert sucker, *Catostomus clarki*, (Rinne and Stefferud, 1996) which comprised 85% of the total number of fish captured in this sample reach. These data suggest aquatic macrohabitat type may be more closely related and important to native fishes in the Verde River than is channel type.

All C-type channels were located immediately (within 100 m) above the E-type sample units on Rio Verde Ranch property. Significantly, a single spikedace, *Meda fulgida*, was taken in sample reach VI. This threatened species is rare (i.e. a threatened species) in the river and is typically taken over gravel/pebble substrates

in run- or LGR-type habitats in the Verde River (Neary et al., 1996; Rinne and Stefferud, 1996). Sample unit VI was classified as a run, and contained almost 75% gravel/pebble substrate. The single E-type channel, run habitat (II) contained a high proportion (60%) of gravel but no pebble substrate. The sand component was greater (36%) in the E-type channel run (II) than the C-type, run habitat (VI; 22 %). These data, similar to that for desert sucker, again suggest aquatic macrohabitat type and associated substrate composition may be more closely related to fish populations than is channel type.

Greater number and biomass of non-natives in the E-type channels seems a paradox. These stable, well-vegetated, undercut bank reaches are desirable in healthy riparian-stream areas (Medina, 1995). Structurally, these E-type channels with undercut banks provide cover for fishes. Presumably, this desirable habitat type should support a greater population of native fish species. However, the two primary fish species inhabiting these streambank areas are smallmouth bass (*Micropterus dolomieu*) and yellow bullhead (*Amerius natalis*), both introduced, predatory species. Preliminary laboratory studies indicate that the former species is a potentially voracious predator on smaller cyprinid species. A single adult bass (>200 mm, total length) has been observed to consume more than 30 red shiners (*Cyprinella lutrensis*) in a 24-hr period. It is reasonable to assume that a similar mechanism of interaction could result with young-of-year or juveniles of any of the six native cypriniform species in the upper Verde. No data are available on competition for space (i.e. undercut banks) with roundtail chub (*Gila robusta*), desert and Sonora (*Catostomus insignis*) suckers, but competition for space also could be a factor limiting both the presence and abundance of native species.

From the stream and fish management viewpoint, it has been assumed that "ideal" E-type well-vegetated channels supporting a greater frequency of undercut banks (Figure 1) are essential for sustaining all of the Verde River's TES fish species. However, the preliminary data indicate that the E-type habitats are also optimal for introduced, predatory species of fishes. Although the native desert sucker may do well in these physical habitats (e.g. sample unit III; Table 1), spinedace apparently do not (Neary et al., 1996; Rinne and Stefferud, 1996). Predation and physical habitat preferences (velocity, substrates, etc.) may explain why spinedace were not found in the "ideal" E-type channels. However, it is not known at this time which process is more important. But, to assume that proliferation of E-type channels in the Verde River will automatically lead to expansion of spinedace populations and the total native fish community could lead to erroneous conclusions about the best approach to their future management.

We can hypothesize that E-type, well-vegetated channels supporting greater frequency of undercut banks are optimal habitat for introduced, predatory species of fishes and may, in part, explain the greater number and biomass of this component of the fish community in the Verde River in these habitat types. Additional data are necessary to confirm or refute this hypothesis. Future sampling of the fish community needs to be conducted at specific sites along the streambank/water interface in order to determine more precisely, the comparative utilization of these habitats by native versus non-native species.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Until we clearly identify the physical habitat preferences and needs of the native TES fishes as well as the impacts of key instream biological processes such as predation and displacement, any conclusions about bank vegetation conditions and the health of a fish community are premature. This preliminary study demonstrated that an "ideal" E-type channel with well-vegetated streambanks does not guarantee the presence of abundant native fishes. We also know that other physical habitat characteristics such as substrate type and stream velocity are also important determinants of good native fish habitat. The importance of aquatic habitats to distribution and abundance of native fishes was demonstrated by the preponderance of desert sucker in the HGR, E-type channel and the presence of the rare spinedace in one C-type channel sample reach.

In southwestern desert rivers such as the Verde, the relationships between

channel morphology and hydrology, well-vegetated streambanks and fish habitat need to be defined further. In turn, the linkage between instream and nearbank habitat and native/non-native fish ratios, density, and biomass needs definition. Because grazing has a direct effect on the vegetation at the terrestrial-aquatic interface, the next step will be to delineate the effects of these impacts to streambank morphology and integrity. Once these impacts are defined, the relationship of livestock grazing on fish populations can be more specifically and validly documented. Ultimately, the levels of grazing or grazing strategies and their relative impacts on physical and biological components of riparian-stream ecosystems can then be determined. Such scientifically-linked information will be vital to sustaining grazing, riparian-stream ecosystems, and native fish communities in low desert rivers.

REFERENCES

- Bevenger, G. S. and R. M. King, 1995. A pebble count procedure for assessing watershed cumulative effects. U.S. Forest Service Research Paper RM-RP-319: 1-17.
- Bjornn, T. C. and D. W. Reiser, 1991. Habitat requirements of salmonids in streams. pages 83-138. In, W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19. Bethesda, Md.
- Medina, A. L., 1995. Native aquatic plants and ecological condition of southwestern wetlands and riparian areas. Pages 329-335. In, D. W. Shaw and D. M. Finch, technical editors. Desired future conditions for southwestern riparian ecosystems: Bringing interest and concerns together. U. S. Forest Service RM-GTR-272. Fort Collins, Colorado.
- Neary, D. G. and A. L. Medina, 1995. Geomorphic response of montane riparian habitat to interactions of ungulates, vegetation and hydrology. Pages 143-147. In, D. W. Shaw and D. M. Finch, technical editors. Desired future conditions for southwestern riparian ecosystems: Bringing interest and concerns together. U. S. Forest Service RM-GTR-272. Fort Collins, Colorado.
- Neary, A. P., J. N. Rinne, and D. G. Neary, 1996. Physical habitat use by spikedace in the upper Verde River, Arizona, pp 23-28. In, Hydrology and Water Resources in Arizona and the Southwest. Vol. 26. Proceedings of the 1996 Meeting of the Arizona Section, American Water Resources Association and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 20, 1996, Tucson, Arizona.
- Platts, W. S., W. F. Megahan, and G. W. Minshall, 1983. Methods for evaluating stream, riparian and biotic conditions. U. S. Forest Service General Technical Report INT-138: 1-70.
- Platts, W. S., et al., 1987. Methods for evaluating riparian habitats with applications to management. U. S. Forest Service General Technical Report INT-221: 1-177.
- Platts, W. S., 1991. Livestock grazing. Pages 389-423. in W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19. Bethesda, Md.
- Rinne, J. N., 1989. Physical habitat use by loach minnow, Tiagroa cobitis (Pisces: Cyprinidae), in southwestern desert streams. Southwestern Naturalist 34(1):109-117.
- Rinne, J. N., 1991. Physical habitat use by spikedace, Meda fulgida in southwestern desert streams with probable habitat competition by red shiner, Notropis lutrensis (Pisces: Cyprinidae). Southwestern Naturalist 36(1):7-13.
- Rinne, J. N., 1992. Physical habitat utilization of fish in a Sonoran Desert stream, Arizona, southwestern United States. Ecology of Freshwater Fishes 1: 1-8.

Rinne, J. N. and J. A. Stefferud, 1996. Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona, pages 13-22. In, Hydrology and Water Resources in Arizona and the Southwest. Vol. 26. Proceedings of the 1996 Meeting of the Arizona Section, American Water Resources Association, and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 20, 1996, Tucson, Arizona.

Rosgen, D. L., 1994. A classification of natural rivers. Catena 22: 169-199.

Stefferud J. A. and J. N. Rinne., 1995. Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes, pp. 26-32. In, Hydrology and Water Resources in Arizona and the Southwest. Volumes 22-25. Proceedings of the 1995 Meeting of the Arizona Section, American Water Resources Association and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 22, 1995, Flagstaff, Arizona.

REFINEMENT OF AQUATIC MACROHABITAT DEFINITION IN THE UPPER VERDE RIVER, ARIZONA

Pamela J. Sponholtz and John N. Rinne¹

Classification of aquatic macrohabitat is important for at least three reasons. First, past observations of stream systems provide insight about how stream habitat changes with flow, sediment deposition, or physical parameters such as depth and velocity. Subjective classification attempts to predict a river's behavior from its appearance (Rosgen 1994). Identification of variables that affect streams can be used to predict responses to disturbances or human-induced changes. If habitat systems are changing on an abiotic level, it can affect biotic constituents (Brown and Moyle 1991).

Second, native fish in the Southwest currently lead a precarious existence, threatened by water diversions, dams, and introduced fishes. As a result, many are formally listed as threatened and endangered (Rinne and Minckley 1991, Minckley and Deacon 1991). Identification of microhabitats utilized by native fish will aid in their conservation and protection. Authors such as Harding (1981) use landform units (mountain ranges, plateaus, and major drainage basins) to characterize stream habitat and fish populations. Bisson et al. (1981) identified three types of riffles and six types of pools in habitats used by salmonids in western Washington. By assigning fish species to guilds based on habitat use, Aadland (1993) was able to determine species composition and relative quality of various habitats. Microhabitats are combined into "mesohabitats" (shallow pool, run, etc.), which makes habitat use by fishes more discernible (Rastetter et al. 1992, Vadas 1992, Vadas and Orth 1997).

Third, collecting accurate data on an entire river system without subdividing its components is infeasible. For example, western aquatic systems are variable with respect to climate, topography, and the resulting hydrology, and identification of macrohabitats will aid in understanding the entire system.

Study of fish populations in the upper Verde

River by the USDA Forest Service, Flagstaff, Arizona commenced in spring of 1994. The main objective is to determine how native fishes persist relative to natural hydrology and introduced fishes (Stefferd and Rinne 1995). A secondary objective is to define aquatic macrohabitats and their use by the total fish community. Preliminary data on habitat definition and delineation were presented by Rinne and Stefferud (1996). More definitive and quantitative, physical descriptions are desirable because qualitative macrohabitat definitions (i.e. runs, glides, high and low gradient riffles) can vary with personnel and level of training (Roper and Scarnecchia 1995, Wang et al. 1996). In the autumn of 1996, five of the seven permanent study reaches in the upper Verde (see Stefferud and Rinne 1995) were sampled more intensively and with additional physical parameters.

The objectives of this paper are to compare the utility of more intensive habitat sampling with previous (1994–1995) less intensive sampling techniques, define aquatic macrohabitats based on quantitative physical data, and estimate the validity of preliminary, qualitative definitions of aquatic macrohabitats provided by Rinne and Stefferud (1996). Objectively defined habitats could then be used to define fish habitat relationships more accurately and validly in the Upper Verde River and perhaps other desert rivers in the Southwest and elsewhere.

Methods

Data were collected within the upper Verde at seven permanently established study reaches (Stefferd and Rinne 1995). Physical habitat data collected in the autumn of 1996 were from only five of the seven sites (Burnt Ranch, FR638, Bear Siding, Perkinsville, and Sycamore Creek). Data from previous years (1994–1995) were collected from all seven study reaches.

The intensity of sampling was increased in 1996 to include transects every 3 meters; in 1994–1995,

¹USDA Forest Service, Rocky Mountain Station, Flagstaff, AZ

three transects were measured irrespective of reach length. We measured transect width and streambank vegetation encroachment. Depth and velocity were measured on three equidistant points along each transect. Four to six transects were established within each habitat type, dependent upon habitat length. In addition, percent gradient at the water surface was measured by a laser survey instrument (Laser Technology Inc.) for each habitat. Substrate was characterized using pebble count methodology where 30 substrate data points were quantified within each sample reach (Bevenger and King 1995). Habitats were subjectively delineated according to our most recent qualitative designations (i.e. spring of 1996). Qualitative definitions used to identify habitats are provided in Table 1. Water velocity was estimated at 0.6 of total depth from the surface with a direct readout current meter (Marsh-McBurney model 2000).

Table 1. Qualitative definitions of general habitat categories used during 1994–1996.

Habitat	Description
Riffle	
High gradient (HGR)	Excessive surface agitation with high velocity; tends to produce standing waves; percent gradient greater than that of low gradient riffles; substrate usually boulders.
Low gradient (LGR)	Broad surface agitation; substrate usually gravel or cobble with interspersed boulders.
Run	Obvious surface velocity with some agitation; depths variable; substrates are typically mosaic of gravel, cobble, and sand.
Glide	Some velocity, but surface generally laminar; depth less than that of pools; substrate usually sand or gravel.
Pool	Little or no surface velocity; generally greater depth than adjacent upstream or downstream habitats; sediments generally gravel, sand, or silt and finer than in adjacent, faster-flowing habitats.

Sampling resulted in 1,567 individual data points for depth and velocity at five sites in 1996. Data points totaled 404 and 502 across seven sites in 1994 and 1995, respectively. We expected that tripling the data points and adding other additional physical parameters, such as gradient, would improve classifications of aquatic macrohabitat. To test this, we used a nested ANOVA to analyze variances among and within sites, and among habitat types. ANOVAs of depth, velocity, and gradient suggested that there was significantly higher variability among transects than among individual points. Thus, data were first averaged across transects before statistics were calculated. Due to unequal sample sizes, a Tukey HSD post hoc test was used to assess differences among habitat types. Data were log transformed to achieve homoscedasticity ($F_{\max}$ test) but they nonetheless retained small deviations from normality. Analyses were performed using SAS version 6.0 (SAS 1984).

We used classification and regression trees (Brieman et al. 1984, Steinberg and Colla 1994) to distinguish patterns in the data for physical variables such as depth, velocity, gradient, and width. This method analyzes all possible predictor variables and divides sample cases according to how well the response classes (habitats) are separated. The analysis first determines the best discriminant boundary value for each variable where data points are most clearly separated from other response classes (Brieman et al. 1984). It then selects the variable that splits sample cases into the purest class memberships. The results are in the form of decision trees and through yes or no questions, new splits are generated until the number of unclassified cases is less than five or all the predictor variables have been used (Verbyla 1987). Each tree is tested by cross-validation and provides an assessment of how consistent the subjective classification is with the quantitative, physical data.

Results

There were generally significant differences ($p < 0.05$) for depth measured in 1994 between and among sites, types, and transects (Tables 2 and 3). Analysis of velocity measurements produced similar results except that there was no significant difference at the transect level. There were no differences between pools and glides for either depth or velocity (Figure 1). Although high and low gradient riffles were not significantly different in depth, they were significantly different in velocity. Runs were significantly greater in depth than riffle habitats and greater in velocity than pools or glides.

Table 2. Mean, standard error, and ranges for depth, velocity, and gradient in habitat surveyed during 1994–1996. Sample sizes are given in parentheses under physical parameters. High gradient riffles are designated as HGR and low gradient riffles as LGR.

	1994 (n = 205)		1995 (n = 251)		1996 (n = 251)		
	Depth (m)	Velocity (m/sec)	Depth (m)	Velocity (m/sec)	Depth (m)	Velocity (m/sec)	% Gradient
HGR	0.34 ± 0.14 0.19 – 0.75	0.86 ± 0.31 0.35 – 1.44	0.35 ± 0.12 0.21 – 0.63	0.63 ± 0.25 0.20 – 1.27	0.24 ± 0.09 0.07 – 0.49	0.55 ± 0.23 0.13 – 1.10	1.83 ± 1.38 0.56 – 4.39
LGR	0.27 ± 0.09 0.09 – 0.55	0.62 ± 0.24 0.27 – 1.28	0.31 ± 0.12 0.10 – 0.63	0.51 ± 0.22 0.20 – 0.98	0.25 ± 0.06 0.07 – 0.49	0.41 ± 0.18 0.11 – 0.87	1.33 ± 0.54 0.72 – 2.38
Run	0.40 ± 0.16 0.13 – 0.98	0.47 ± 0.21 0.10 – 0.97	0.43 ± 0.16 0.21 – 0.91	0.41 ± 0.16 0.11 – 0.70	0.31 ± 0.07 0.13 – 0.49	0.35 ± 0.12 0.08 – 0.56	0.70 ± 0.77 0.04 – 2.31
Glide	0.54 ± 0.21 0.16 – 1.13	0.21 ± 0.09 0.03 – 0.57	0.44 ± 0.23 0.13 – 1.20	0.19 ± 0.09 0.00 – 0.35	0.34 ± 0.15 0.08 – 0.78	0.27 ± 0.14 0.05 – 0.62	0.50 ± 0.28 0.07 – 1.15
Pool	0.61 ± 0.22 0.21 – 1.20	0.29 ± 0.25 0.00 – 0.96	0.71 ± 0.32 0.33 – 1.79	0.21 ± 0.16 0.00 – 0.82	0.49 ± 0.22 0.12 – 1.05	0.18 ± 0.19 0.00 – 0.77	0.49 ± 0.42 0.00 – 2.00

Table 3. Nested ANOVA results for 1994–1996. Sample sizes are designated by N and site x type interactions are designated site-type. Nonsignificant values at the p = 0.05 level are in bold type.

	N	Site	Type	Site-Type	Transect
1994					
Depth	205	F = 7.50 p < 0.01	F = 40.64 p < 0.01	F = 4.06 p < 0.01	F = 1.73 p < 0.01
Velocity	205	F = 15.69 p < 0.01	F = 79.45 p < 0.01	F = 2.69 p < 0.05	F = 1.33 p = 0.05
1995					
Depth	251	F = 1.87 p = 0.09	F = 20.39 p < 0.01	F = 1.05 p = 0.41	F = 3.03 p < 0.01
Velocity	251	F = 14.03 p < 0.01	F = 58.24 p < 0.01	F = 3.15 p < 0.01	F = 0.54 p < 0.99
1996					
Depth	327	F = 8.80 p < 0.01	F = 43.46 p < 0.01	F = 8.80 p < 0.01	F = 2.24 p < 0.01
Velocity	319	F = 28.73 p < 0.01	F = 84.23 p < 0.01	F = 9.41 p < 0.01	F = 2.26 p < 0.01
Gradient	61	F = 157.20 p < 0.01	F = 244.78 p < 0.01	F = 54.60 p < 0.01	F = 1.10 p < 0.01

Subjective classification of habitat matched definitions based on classification trees less than 50 percent of the time, except for 70.8 percent agreement on high gradient riffles (Figure 2). Glide habitats were classified as pools 14 out of 40 times. Low gradient riffles were classified as high gradient riffles 13 out of 44 times and as runs 18 out of 44 times. Subjective classification matched classification tree definitions only 45.4 percent of the time for 1994 (Table 4). The important variables used in the tree analyses were velocity followed by depth.

Depth was significantly different among habitats and transects in 1995. Velocity was significantly different among sites and habitats (Table 3). Post hoc tests indicated that pools are the only habitats that are significantly deeper from the other types (Figure 3). Although pools had the lowest velocity, this habitat type was not significantly different in velocity from glides. Similarly, velocities for high and low gradient riffles were not significantly different. Classification tree analysis indicated that glides were misclassified as pools 12 out of 50 times. Low gradient riffles were correctly classified only 7 out of 39 times (17.9%) and were misclassified as high gradient riffles 16 out of 39 times (Figure 2). Habitats were correctly classified 53 percent of the time (Table 4). As in 1994, the important variables used to generate the classification tree were velocity followed by depth.

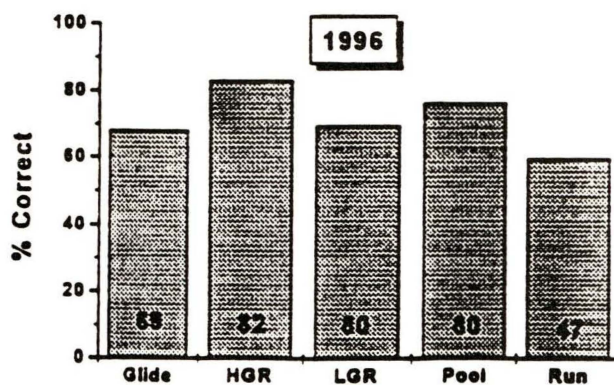
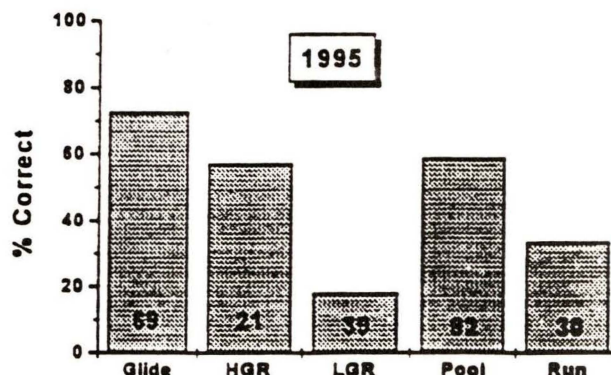
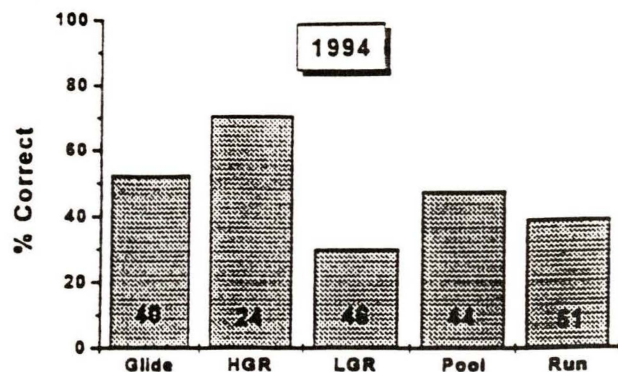
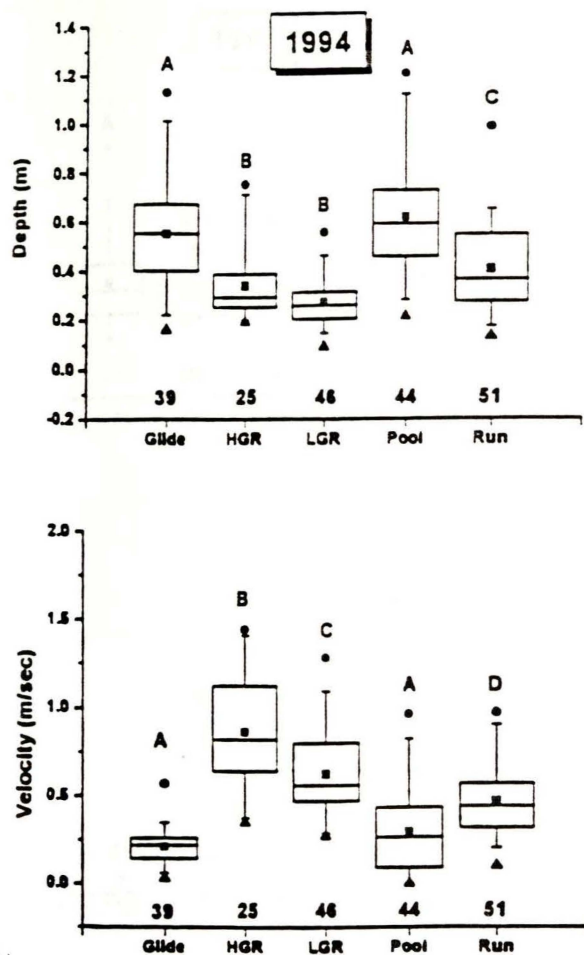


Figure 2. Percent matching objective, subjective classifications for habitats in 1994–1996. Sample sizes are given at the base of each column.

est velocities and gradients. Correct classifications of individual habitats by classification tree analysis were all above 50 percent with a total of 72.8 percent matched (Figure 2). In general, glides were still misclassified as pools (17 out of 69 times) and low gradient riffles were misclassified as runs and high gradient riffles (Table 4). Finally, the most important variable in 1996 was gradient followed by velocity and depth.

Figure 1. Box plots of depth (m) and velocity (m/sec) for 1994. Plots show minimum ($\blacktriangle$), maximum ($\bullet$), and mean ($\blacksquare$) values. The box lines show, from the top, the 75th median and 25th quantiles. Differences between habitats were determined using Tukey's HSD test at the $p = 0.05$ significance level. Significantly different habitats are designated by different letters. Sample sizes (transects) are provided below the plots.

Depth, velocity, and gradient measurements completed in 1996 were significantly different across sites, types, and transects (Table 3). However, post hoc tests indicated that glides, runs, and pools are deeper than riffles, but glides were not distinguishable from either pools or runs, which were different (Figure 4). All habitat types had significantly different velocities except runs and low gradient riffles. Gradient measurements were significantly different across all types. In general, pools were the deepest and had the lowest velocities and gradients. In contrast, high and low gradient riffles were the shallowest and had the high-

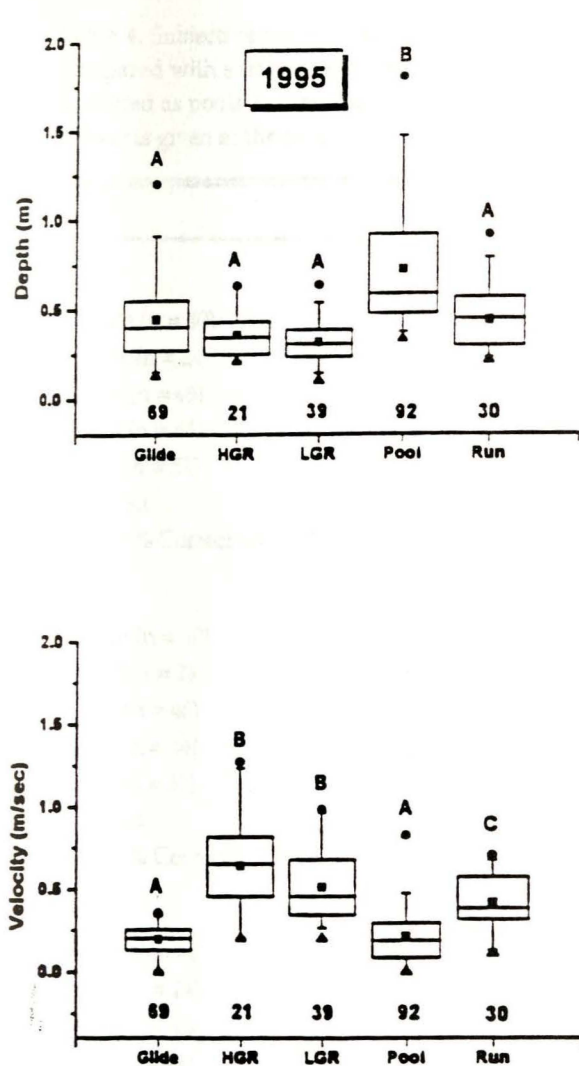


Figure 3. Box plots of depth (m) and velocity (m/sec) for 1995. Otherwise as for Figure 1.

Discussion

Depth and velocity measurements were not specific enough to detect subtle differences between the two riffle types. Depth was not significantly different between high and low gradient riffles in any of the three years, despite 30 percent more transects being sampled in 1996. In most cases, velocity was also not significantly different between high and low gradient riffles. This was also the case for pools and glides. Gradient, added in 1996, was the only measurement that clearly defined habitat types. Nonetheless, combined with gradient measurements, depth and velocity may remain powerful tools for habitat classification. To illustrate this further, a classification tree using gradient, velocity, and depth as the defining habitat parameters gives a correct classification 82 percent of the time. When gradient is used by itself with

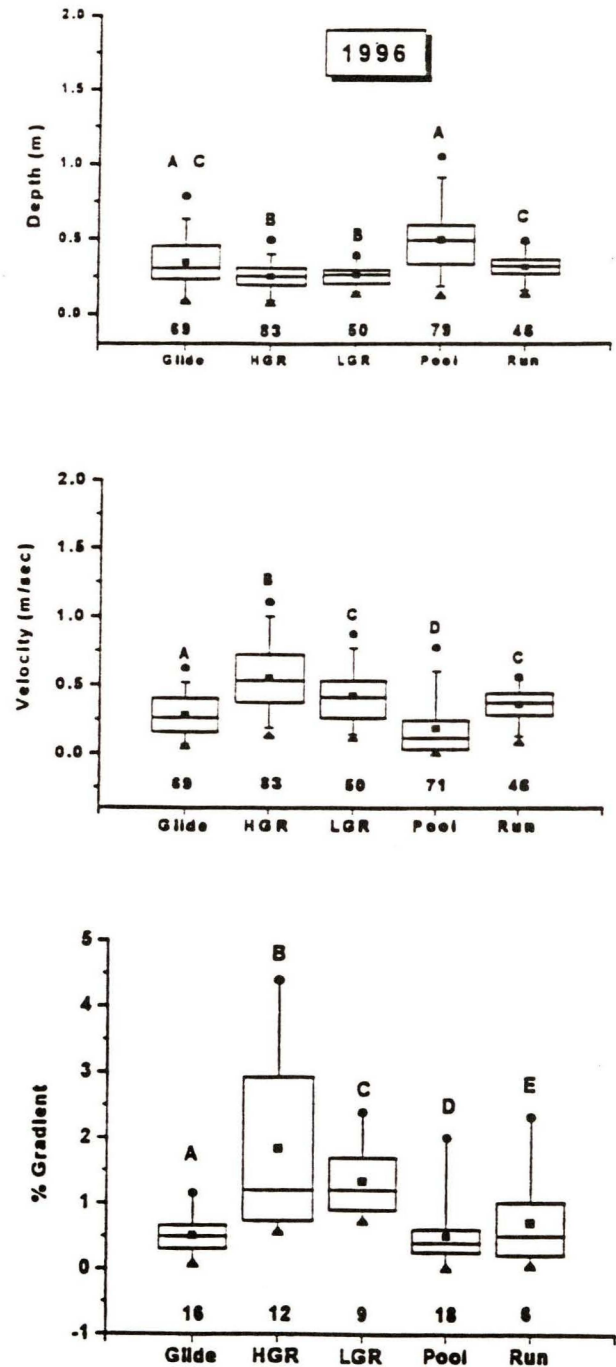


Figure 4. Box plots of depth (m), velocity (m/sec), and percent gradient for 1996. Otherwise as for Figures 1 and 3.

the same data, the percent correctly classified is 71.9 percent. Although further study needs to be completed before these trees can be accepted, gradient measurements should be an integral part of habitat classification.

Generally, sites have significantly different depth, velocity, and gradient. Although not surprising considering the variability of a natural river system, habitats may be site specific. That is,

Table 4. Subjective vs. objective classifications of habitats for 1994–1996. Objective categories (vertical column) are compared with subjective (horizontal row) classifications. For example, glides were classified as glides 21 times but classified as pools 14 times and as runs 5 times. Percent correct classification for each habitat and the overall percent correct is given at the bottom of the table.

	Glide	HGR	LGR	Pool	Run
1994					
Glide (n = 40)	21	0	0	14	5
HGR (n = 24)	0	17	5	0	2
LGR (n = 46)	0	13	14	1	18
Pool (n = 44)	16	4	1	21	2
Run (n = 51)	7	5	15	4	20
Correct	0.525	0.708	0.304	0.477	0.392
Total % Correct = 0.454					
1995					
Glide (n = 40)	50	3	3	12	1
HGR (n = 24)	2	12	3	1	3
LGR (n = 46)	8	16	7	1	7
Pool (n = 44)	22	7	3	54	6
Run (n = 51)	7	7	6	0	10
Correct	0.725	0.571	0.179	0.587	0.333
Total % Correct = 0.530					
1996					
Glide (n = 40)	47	0	2	17	3
HGR (n = 24)	1	68	10	2	1
LGR (n = 46)	0	9	34	0	6
Pool (n = 44)	10	2	1	61	8
Run (n = 51)	9	1	6	3	28
Correct	0.681	0.829	0.694	0.763	0.596
Total % Correct = 0.728					

a glide at Burnt Ranch may have different ranges of physical parameters than a glide at the Sycamore site. This may be due, in part, to discharge. Base flows at the Paulden gauge range from 0.7 to 3.7 m³/sec (average 1.3 m³/sec), whereas the downstream gauge at Clarkdale records flows three times greater from 2.2 to 14.7 m³/sec (average 5.7 m³/sec; Smith et al. 1993). It is possible that a glide directly below Paulden is going to be shallower and have less velocity than a glide near Sycamore. Although both habitats are considered glides, they may differ with respect to physical characteristics due to discharge.

Classification trees clearly indicated that subjective classification is highly variable and unreliable. We assumed that increasing the number of meas-

urements within a sample would more clearly define macrohabitats. This assumption appears incorrect. Although 1996 habitats were misclassified less often than 1994 or 1995, the increased accuracy of prediction was most likely due to the addition of gradient as a variable, rather than increased sample size. Ranking of important classification variables supports this speculation. That is, in 1996, gradient replaced depth and velocity as the top ranking variable used in classification.

At this point, interpretation of the classification trees should be approached cautiously. The models generated for analysis of aquatic macrohabitats in the upper Verde River are not predictive. These models were used to compare only the data that had already been collected to a logical pattern of

habitat classification. Also, our model assumed an equal cost if stream habitats were misclassified. However, classifying a high gradient riffle as a pool is more erroneous than classifying it as a low gradient riffle. Thus, in future models, decisions should be weighted according to the cost of being misclassified.

Conclusions

We demonstrated that gradient was a valuable tool for aquatic macrohabitat classification, but the procedure for obtaining gradients needs to be standardized among field personnel. Reducing the variability within gradient measurements (increasing replication) will provide more refined habitat classification. When performed alone, depth and velocity measurements can provide distinction between some habitats. However, when these parameters are combined with gradient analyses, they provide a powerful tool capable of finer habitat classification. Also, the subjective classifications are flawed. Although some habitats are easily distinguished from one another (e.g. pools from high gradient riffles), some habitats are difficult to subjectively categorize based on a few physical measurements. Thus, for objective and reliable definition of macrohabitats, classification should be done after the data are analyzed and patterns can be established. Finally, additional analysis of habitats throughout the Verde River and perhaps in other rivers is essential to determine how variables change with stream flow.

Acknowledgments

Support for this project was provided by the Arizona Heritage Foundation. The authors wish to thank both Jerry Stefferud for his advice and Rudy King for his extensive statistical assistance.

References Cited

- Aadland, L.P. 1993. Stream habitat types: Their fish assemblages and relationship to flow. *North American Journal of Fisheries Management* 13: 790-806.
- Bevenger, G.S. and R.M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. *USDA Forest Service Research Paper RM-319*, pp. 1-17.
- Bisson, P.A., J.L. Nielsen, R.A. Palmason, and L.E. Grove. 1981. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. Pages 62-73 in *Proceedings, Acquisition and utilization of aquatic habitat inventory information*, October 28-30, Portland, OR.
- Breiman, L., J.H. Friedman, R.A. Olshen, and C.J. Stone. 1984. *Classification and regression trees*. Wadsworth, Inc., Belmont, CA.
- Brown, L.R. and P.B. Moyle. 1991. Changes in habitat and microhabitat partitioning within an assemblage of stream fishes in response to predation by Sacramento squawfish (*Ptychocheilus grandis*). *Canadian Journal of Fisheries and Aquatic Sciences* 48: 849-856.
- Harding, E.A. 1981. Landform subdivisions as an interpretive tool for stream and fish habitat assessment. Pages 41-46 in *Proceedings, Acquisition and utilization of aquatic habitat inventory information*, October 28-30, Portland, OR.
- Minckley, W.L. and J.E. Deacon, editors. 1991. *Battle against extinction*. University of Arizona Press, Tucson. 517 pp.
- Rastetter, E.B., A.W. King, B.J. Cosby, G.M. Hornberger, R.V. O'Neill, and J.E. Hobbie. 1992. Aggregating fine-scale ecological knowledge to model coarser-scale attributes of ecosystems. *Ecol. Applic.* 2: 55-70.
- Rinne, J.N. and W.L. Minckley. 1991. Native fishes of arid lands: A dwindling natural resource of the desert Southwest. *USDA Forest Service Gen. Tech. Rep. RM-206*, pp. 1-45. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
- Rinne, J.N. and J.A. Stefferud. 1996. Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona. Pages 13-22 in *Hydrology and Water Resources in Arizona and the Southwest*, Vol. 26, *Proceedings of the 1996 meetings of the Arizona Section, American Water Resource Association and the Hydrology Section, Arizona-Nevada Academy of Sciences*, April 20, Tucson, AZ.
- Roper, B.B. and D.L. Scarnecchia. 1995. Observer variability in classifying habitat types in stream surveys. *North American Journal of Fisheries Management* 15: 49-53.
- Rosgen, D.L. 1994. A classification of natural rivers. *Catena* 22: 169-199.

- SAS 1984. User's Guide: Statistics. Version 5. SAS Institute, Inc., Cary NC. 956 pp.
- Smith, C.F., P.D. Rigas, L.K. Ham, N.R. Duet, and D.W. Anning. 1993. Water resources data, Arizona, Water Year 1993. U.S. Geological Survey Water-Data Report AZ-93-1, Phoenix.
- Stefferdud J.A. and J.N. Rinne. 1995. Preliminary observations on the sustainability of fishes in a desert river: The roles of streamflow and introduced fishes. In *Hydrology and Water Resources in Arizona and the Southwest*, Vols. 22-25, pp. 26-32. Proceedings of the Arizona Section, American Water Resource Association and the Hydrology Section, Arizona-Nevada Academy of Sciences, April 22, 1995, Flagstaff, AZ.
- Steinberg, D. and P. Colla. 1994. CART for Windows. Salford Systems, San Diego, CA.
- Vadas, R.L., Jr. 1992. Seasonal habitat use, species associations, and assemblage structure of forage fishes in Goose Creek, northern Virginia. II. Mesohabitat patterns. *J. Freshwater Ecol.* 7: 149-164.
- Vadas, R.L., Jr. and D.J. Orth. 1997. Species associations and habitat use of stream fishes: The effects of unaggregated-data analysis. *J. Freshwater Ecol.* 12: 27-37.
- Verbyla, D.L. 1987. Classification trees: A new discrimination tool. *Canadian Journal of Forestry Research* 17: 1150-1152.
- Wang, L., T.D. Simonson, and J. Lyons. 1996. Accuracy and precision of selected stream habitat estimates. *North American Journal of Fisheries Management* 16: 340-347.

SUSTAINABILITY OF FISHES IN DESERT RIVER: PRELIMINARY OBSERVATIONS ON THE ROLES OF STREAMFLOW AND INTRODUCED FISHES

Jerome A. Stefferud¹ and John N. Rinne²

Patterns of stream discharge and the presence of introduced fishes appear to interact to influence community dynamics and sustainability of native fishes in streams of the arid Southwest (Minckley and Meffe 1987). A long history of water development and diversion coupled with numerous introductions of non-native and exotic fishes has resulted in few streams in Arizona retaining predominantly native fish communities (Miller 1960; Minckley and Douglas 1991). Although native fishes yet occur in most drainages in the state, the number of species occurring in any stream typically is reduced from historic levels. Where native fishes are present, the community dynamics apparently are altered by non-native or exotic species (Minckley and Deacon 1968; Rinne and Minckley 1991; Deacon and Minckley 1991).

Minckley and Meffe (1987) contrasted the differences in flood hydrology between southwestern arid lands and those of lowland mesic regions of the central and eastern U.S. They found that most of the annual water yield in southwestern systems was produced during high discharges for brief periods of time, whereas low discharges produced a far greater proportion of total yield from mesic systems. The differential effects of these patterns on fishes have had a significant impact on the native-introduced mix of species now present in the Southwest. Minckley and Meffe (1987) concluded that native southwestern fishes were better adapted to withstanding the effects of large floods than non-native fishes. During significant floods, non-native species were either displaced or destroyed, whereas native species maintained position in or adjacent to channel habitats, persisted in microrefuges, or rapidly recolonized if displaced.

The upper Verde River, central Arizona, from its source at Sullivan Lake to the mouth of Sycamore Creek 60 km downstream, is rare among Arizona rivers in that it still retains its native fish fauna. The six native fishes historically recorded there still occur abundantly, and at least seven non-native species are established, with many others reported. The native fish fauna includes longfin dace (*Agosia chrysogaster*), speckled dace (*Rhinichthys osculus*), roundtail chub (*Gila robusta*), spinedace (*Meda fulgida*), desert sucker (*Pantosteus clarki*), and Sonora sucker (*Catostomus insignis*) (Minckley 1973; Minckley 1985). The skeletal remains of razorback sucker (*Xyrauchen texanus*) and Colorado squawfish (*Ptychocheilus lucius*) were found at an archaeological site near Perkinsville (dated ca. 1300–1400 A.D.) (Minckley and Alger 1968), and both species have recently been stocked in this reach (Hendrickson 1993). Non-native species commonly found there include red shiner (*Cyprinella lutrensis*), fathead minnow (*Pimephales promelas*), common carp (*Cyprinus carpio*), green sunfish (*Lepomis macrochirus*), smallmouth bass (*Micropterus dolomieu*), channel catfish (*Ictalurus punctatus*), and yellow bullhead (*Ameiurus natalis*) (Minckley 1973; Minckley 1985; Hendrickson 1993). Reasons for retention of the native fish fauna are unclear, but may be related to the canyon-bound nature of the river and its relatively unregulated flows that can scour the channel and evidently affect the dynamics of the fish community (Minckley 1985; Sullivan and Richardson 1992).

Many studies of the fishes and aquatic ecology of the upper Verde River are available in the published literature and in scattered agency reports, but none have investigated the long-term interrelationships of the fish community with disturbance events. In order to gain an understanding of how the fish community in the upper Verde River responds to disturbance events (drought, flooding, invasions by non-native species), we initiated a study in 1994 to

¹U.S. Dept. of Agriculture, Forest Service, Tonto National Forest, Phoenix AZ

²U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Flagstaff AZ

observe changes in species abundance and occurrence through time and space, fish densities and biomass relative to aquatic habitat types, and mobility of fishes. Long-term and consistent monitoring of the fishery at several sites along the length of the upper Verde River will be accomplished annually during a 10-year period. The objective of the study is to determine the roles and relative influence of physical (hydrology and geomorphology) and biological (introduced fish population dynamics, fish mobility, native versus non-native interactions, reintroduction of native species) factors in the sustainability of native fishes in time (10 years) and through space (7 sites over 60 km). Our purpose herein is to report preliminary results from our first year of sampling.

Methods

The general location of study sites was chosen based primarily on reasonable access and linear disposition over the 60-km reach. Specific locations were based on channel morphology and types of habitat present. Access to this reach of the Verde River, in particular the 20 km east of Perkinsville, is limited by the topography of the watershed and lack of roads. There is only one bridge across the river (at Perkinsville), and access to the valley bottom at other sites is limited to four-wheel-drive vehicles or foot trail. The canyon bottom is administratively restricted by the Forest Service to non-motorized travel, although the Topeka and Santa Fe Railroad parallels the river downstream from Perkinsville. Seven sites were chosen starting at a point 7.6 km east of Sullivan Lake and ending at the confluence of Sycamore Creek (Figure 1). Sites were linearly separated by distances of 3.3 to 16.2 km.

We proposed to sample 250 to 300 m of stream at each site, but exact length was dependent on habitat complexity at the site. In general, a riffle-pool sequence was selected, and then the study site was expanded to include other significant habitat types present. Habitat types were defined by combinations of gradient, subjective descriptions of channel shape and turbulence, and relative position in the stream channel (Bisson et al. 1982; Aadland 1993).

Habitat types were sampled for fish in sequence, progressing from downstream to upstream. The methods used were appropriate to the habitats sampled, including backpack DC

electrofishing units, seines (3.2 mm mesh), dip-nets, and trammel nets. All fish captured within a habitat type were identified to species and enumerated. Measurements of total length (TL) were obtained from a subsample of each species. Fish were then returned alive to the stream.

Each habitat type was numbered in sequence and named (lateral scour pool, mid-channel pool, backwater pool, glide, run, low and high gradient riffles, and edgewater). Length, width, depth, velocity, and substrate for each habitat type were recorded. For future reference, a map of the entire site was sketched and location of habitat types noted. Water chemistry was recorded, including dissolved oxygen, conductivity, total dissolved solids, pH, turbidity, and temperature. Total length of the site was measured, and notes on the shape of the valley bottom, width of the active channel, and land uses were made.

Results

Sixty-seven habitat types at seven sites were sampled for fish and measured for habitat characteristics during April and May 1994. In October 1994, 12 selected habitat types at Burnt Ranch and Sycamore sites were resampled for fish. Thirteen species (8,705 individuals) were captured during the spring sampling (Table 1), and 11 species (1,148 individuals) in the autumn (Table 2). The total number of individuals captured per site ranged from 451 to 2,837. The number of native species comprised 82 percent of the spring sample, but only 54 percent of the autumn sample. The non-native red shiner accounted for 17 percent of the total spring fish catch, and 39 percent of the autumn. Seining accounted for 57 percent, and electrofishing 43 percent, of the total catch in spring. Only seining was used during autumn sampling.

The relative abundance of native and non-native species varied longitudinally and seasonally. Longfin dace was common at the upper two sites but uncommon or absent at the lower sites, whereas the opposite pattern applied to speckled dace. Roundtail chub was relatively common at the lower four sites in the spring, but was found in greater abundance at the upper site in the autumn. The threatened spikedace was abundant at the upper site, extremely rare at the next three sites downstream, and relatively common at the lower three sites during spring sampling. However, it was absent from

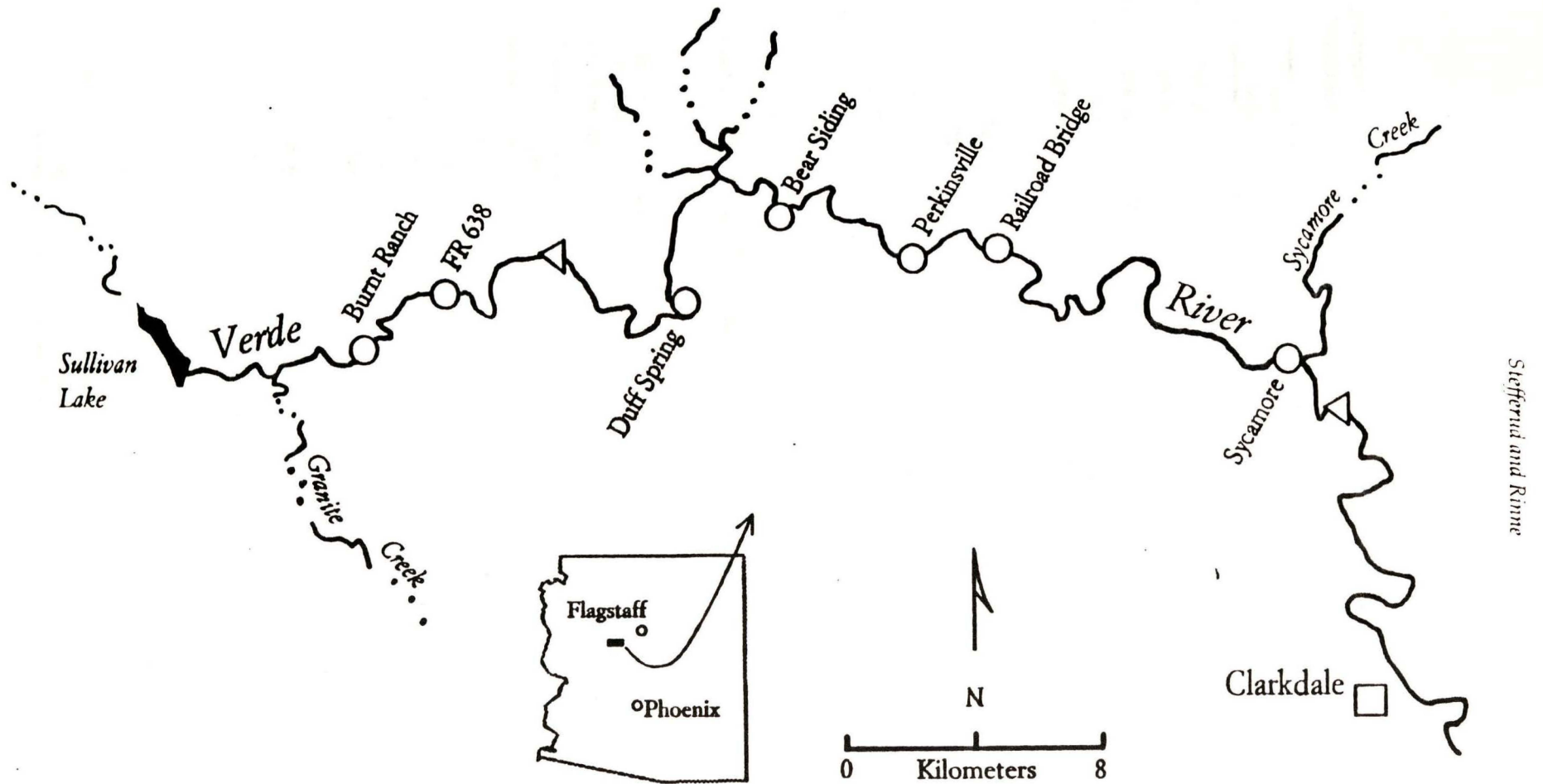


Figure 1. Map of study area with open circles indicating sample sites. Triangles indicate locations of USGS Paulden and Clarkdale gages.

Table 1. Summary of fish collections from Upper Verde River, April–May 1994 (number and relative abundance as percentage).

Species	Burnt Ranch	FR638	Duff Spring	Bear Siding	Perkinsville	Railroad Bridge	Sycamore	Total
Longfin dace	1072 (53.5)	227 (32.7)	--	--	19 (0.7)	--	1 (0.1)	1319 (15.2)
Desert sucker	339 (16.9)	230 (33.1)	192 (32.5)	382 (31.1)	885 (31.2)	237 (52.5)	379 (42.2)	2644 (30.4)
Sonora sucker	278 (13.9)	154 (22.2)	329 (55.8)	357 (29.1)	442 (15.6)	27 (6.0)	223 (24.8)	1810 (20.8)
Roundtail chub	15 (0.7)	18 (2.6)	28 (4.7)	249 (20.3)	244 (8.6)	57 (12.6)	165 (18.4)	776 (8.9)
Spike dace	257 (12.8)	2 (0.3)	1 (0.2)	1 (0.1)	37 (1.3)	38 (8.4)	92 (10.2)	428 (4.9)
Speckled dace	--	--	--	3 (0.2)	61 (2.2)	88 (19.5)	19 (2.1)	171 (2.0)
Yellow bullhead	2 (0.1)	3 (0.4)	1 (0.2)	1 (0.1)	12 (0.4)	2 (0.4)	10 (1.1)	31 (0.4)
Common carp	1 (T)	--	3 (0.5)	4 (0.3)	11 (0.4)	--	4 (0.4)	23 (0.3)
Red shiner	39 (1.9)	61 (8.8)	32 (5.4)	227 (18.5)	1109 (39.1)	2 (0.4)	3 (0.3)	1473 (16.9)
Channel catfish	--	--	--	1 (0.1)	4 (0.1)	--	--	5 (0.1)
Green sunfish	--	--	--	--	4 (0.1)	--	--	5(T)
Smallmouth bass	2 (0.1)	--	4 (0.7)	3 (0.1)	2 (0.1)	--	3 (0.3)	14 (0.2)
Fathead minnow	--	--	--	--	7 (0.2)	--	--	7 (0.1)
N	2005	695	590	1228	2837	451	899	8705

Table 2. Summary of fish collections from Upper Verde River, October 1994 (number and relative abundance as percentage, in parentheses).

Species	Burnt Ranch	Sycamore	Total
Longfin dace	94 (15.5)	--	94 (8.2)
Desert sucker	31 (5.1)	93 (17.2)	124 (10.8)
Sonora sucker	214 (35.3)	25 (4.6)	239 (20.8)
Roundtail chub	50 (8.2)	17 (3.1)	67 (5.8)
Spike dace	93 (15.3)	--	93 (8.1)
Speckled dace	--	1 (0.2)	1 (T)
Yellow bullhead	1 (0.2)	6 (1.1)	7 (0.6)
Common carp	67 (8.2)	1 (0.2)	68 (5.9)
Red shiner	50 (8.2)	395 (73.0)	448 (38.8)
Channel catfish	--	--	--
Green sunfish	--	--	--
Smallmouth bass	2 (0.3)	3 (0.6)	5 (0.4)
Fathead minnow	5 (0.8)	--	5 (0.4)
N	607	541	1148

the lower site during autumn sampling, whereas red shiner comprised the majority of the autumn sample at that site.

Roundtail chubs in October ($n = 67$; 98 to 420 mm TL; $\bar{x} = 110.2$ mm TL) were significantly greater in length ($t = -12.6372$; $P < 0.05$) than those in spring ($n = 456$; 38 to 433 mm TL; $\bar{x} = 99.5$ mm TL).

Length of sites sampled during spring ranged from 124 to 361 m, and the amount of surface area per site ranged from 1,185 to 2,279 m². In autumn the length of sites sampled was 158 to 270 m and surface area was 993 to 439 m². Among these sites, 67 habitat types were sampled in spring and 12 in autumn. Habitat types were predominantly riffles (36%), followed by runs (25%), glides (16%), lateral scour pools (15%), and backwater pools (4%). Abundance of fish was highest in backwater pools, and was mostly attributable to the presence of Sonora sucker and roundtail chub (Figure 2). Longfin dace and spikedace were only in riffle, run, or glide habitats, and speckled dace were primarily

in riffle habitats. Desert sucker were in all habitat types. Red shiners were in glide and run habitats.

Discussion

Baseflow in the upper Verde River is primarily derived from springs and subsurface input; there are few tributaries and most are intermittent. Stream discharge is relatively constant at a site and is significantly affected only during extreme runoff or droughts. At the Paulden gage annual mean discharge is $1.3 \text{ m}^3/\text{sec}$, with monthly means between 0.7 and $3.7 \text{ m}^3/\text{sec}$. Downstream at the Clarkdale gage annual mean discharge is $5.7 \text{ m}^3/\text{sec}$, and monthly means are between 2.2 and $14.7 \text{ m}^3/\text{sec}$ (Smith et al. 1993).

As is typical of desert streams and rivers, flooding can be significant in the Verde River (Figure 3). During January–February 1993 at Paulden, a peak discharge of $657 \text{ m}^3/\text{sec}$ was preceded by three floods that each exceeded $200 \text{ m}^3/\text{sec}$. A peak discharge of $1,500 \text{ m}^3/\text{sec}$ at Clarkdale was also preceded by three floods, each exceeding $500 \text{ m}^3/\text{sec}$ (Smith et al. 1993). The peak floods, estimated to have recurrence intervals of 50 to 60 years (U.S. Geological Survey 1992), “reset” the biological and geomorphological reference points baseline. That is, riparian vegetational succession was set back to a base level, channel morphology was modified, and stream bed materials were sorted and rearranged.

The floods probably benefited native fish by breaking up embedded bottom materials (Mueller 1984) and reducing the populations of non-native fishes (Minckley and Meffe 1987). Observations in summer 1993 indicated that roundtail chub, spikedace, and desert and Sonora sucker had high reproductive success that year. Our collections in 1994 showed a strong 1 year age class for those species. With the exception of red shiner, non-native fishes comprised less than 1 percent of the total catch, although they may be under-represented because our sampling methods do not adequately sample habitats (e.g., deep pools) where non-native predators are more likely to occur. Red shiners were present at all sites but were common or abundant only at the middle two sites, and were a minor component of the fish fauna at the others.

During February 1995, the Verde River expe-

rienced floods in the 8 to 10 year recurrence intervals. Casual observations at sampling sites on March 29–30 under extremely clear water suggested a drastic reduction in all fishes in the sampling reaches. Although our study is focused on the effects of disturbances to the system, we did not anticipate record floods occurring during the initial stages of the project. Results of sampling this year and during out-years will make an interesting comparison with our present knowledge of the system.

Literature Cited

- Aadland, L.P. 1993. Stream habitat types: Their fish assemblages and relationship to flow. *North American Journal of Fisheries Management* 13:790–806.
- Bisson, P.A., J.L. Nielsen, R.A. Palmasson, and L.E. Grove. 1982. A system of naming habitat types in small streams with examples of habitat utilization by salmonids during low streamflow. In N.B. Armantrout (editor). *Acquisition and utilization of aquatic habitat inventory information*, pp. 62–73. *Proceedings of a symposium held 28–30 October 1981, Portland, Oregon*. American Fisheries Society, Western Division, Bethesda, MD.
- Deacon, J.E., and W.L. Minckley. 1991. Western fishes and the real world: The enigma of “endangered species” revisited. In W.L. Minckley and J.E. Deacon (editors). *Battle against extinction*, pp. 405–413. The University of Arizona Press, Tucson.
- Hendrickson, D.A. 1993. Evaluation of the razorback sucker (*Xyrauchen texanus*) and Colorado squawfish (*Ptychocheilus lucius*) reintroduction programs in central Arizona based on surveys of fish populations in the Salt and Verde rivers from 1986 to 1990. Arizona Game and Fish Department, Phoenix.
- Miller, R.R. 1960. Man and the changing fish fauna of the American southwest. *Papers of the Michigan Academy of Science, Arts, and Letters* 46:365–404.
- Minckley, W.L. 1973. *Fishes of Arizona*. Arizona Game and Fish Department, Phoenix.
- Minckley, W.L. 1985. Native fishes and natural aquatic habitats in U.S. Fish and Wildlife Service Region II west of the continental divide. Arizona State University, Tempe, for U.S. Fish and Wildlife Service, Albuquerque, New Mexico.
- Minckley, W.L., and N.T. Alger. 1968. Fish remains from an archaeological site along the Verde River, Yavapai County, Arizona. *Plateau* 40:91–97.
- Minckley, W.L., and J.E. Deacon. 1968. Southwestern fishes and the enigma of “Endangered species.” *Science* 159:1424–1432.

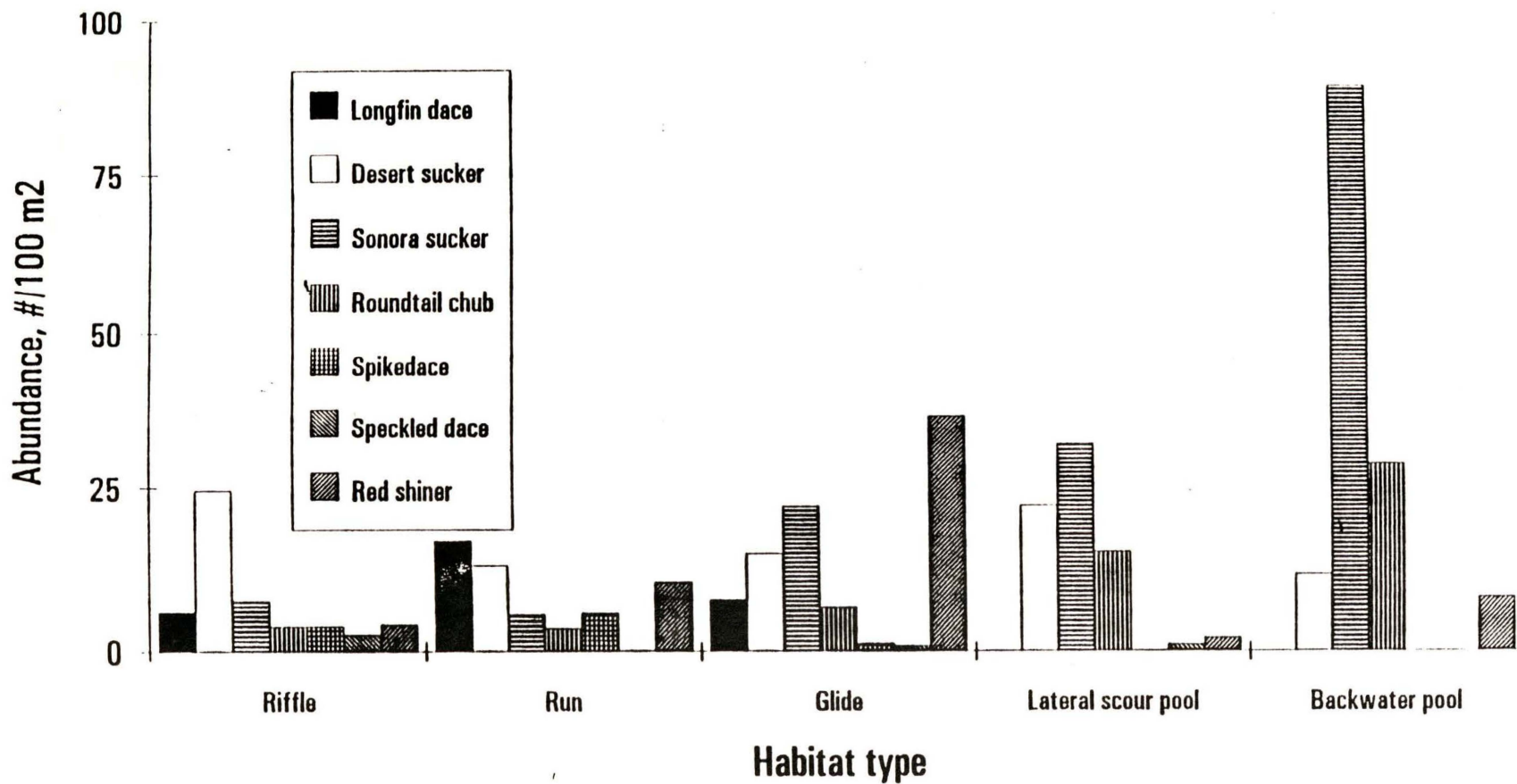


Figure 2. Abundance (number of individuals per 100 m² of surface area sampled) of selected fishes in major habitat types in Verde River captured during sampling in spring and autumn, 1993.

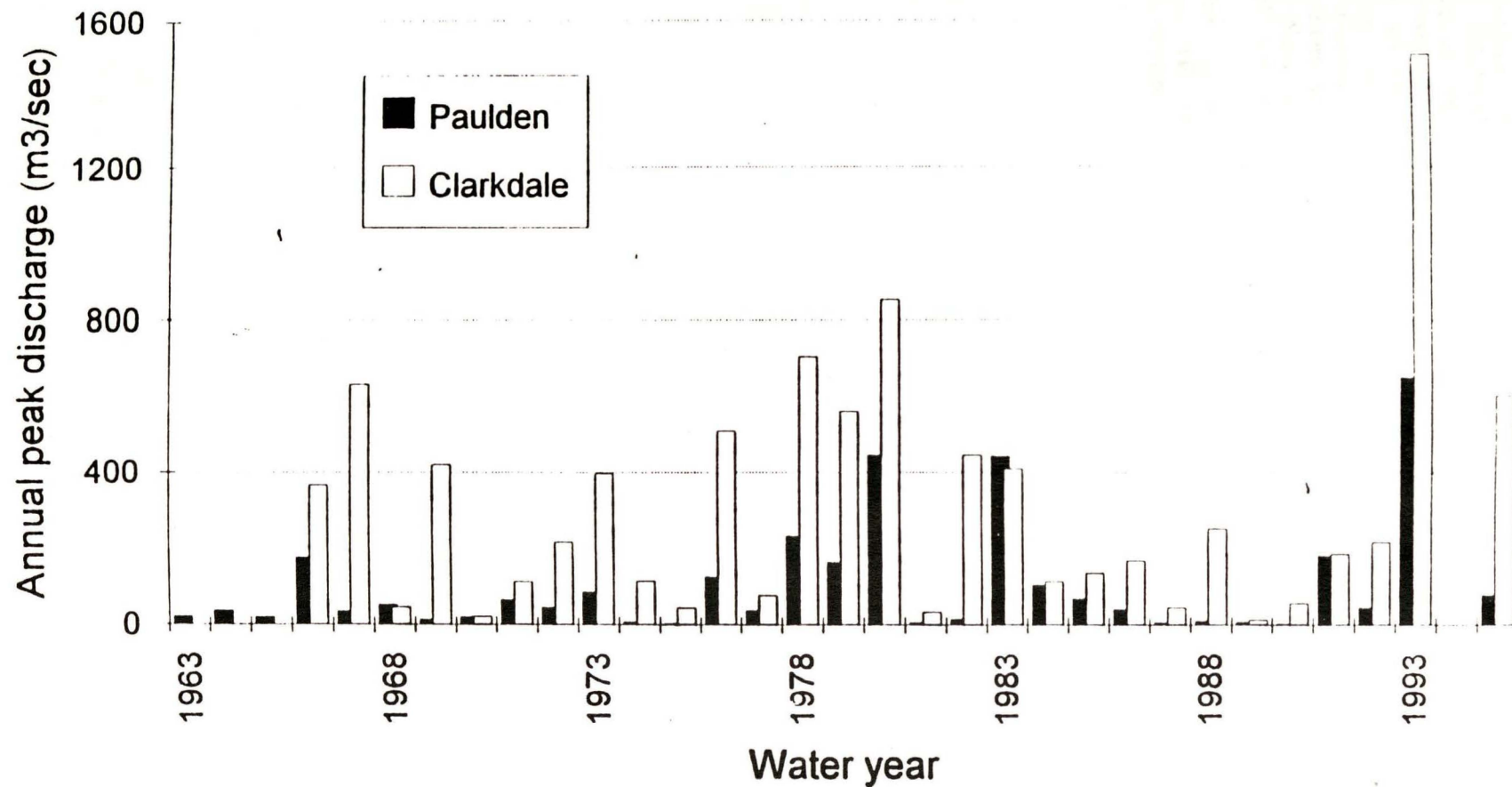


Figure 3. Annual peak discharge at Paulden and Clarkdale gages during water years 1963 to 1995 (U.S. Geological Survey records).

- Minckley, W.L., and M.E. Douglas. 1991. Discovery and extinction of western fishes: A blink of the eye in geologic time. In W.L. Minckley and J.E. Deacon (editors). *Battle against extinction*, pp. 7-17. The University of Arizona Press, Tucson.
- Minckley, W.L., and G.K. Meffe. 1987. Differential selection by flooding in stream fish communities of the arid American southwest. In W.J. Matthews and D.C. Heins (editors). *Community and evolutionary ecology of North American stream fishes*, pp. 93-104. University of Oklahoma Press, Norman.
- Mueller, G. 1984. Spawning by *Rhinichthys osculus* (Cyprinidae) in the San Francisco River, New Mexico. *The Southwestern Naturalist* 29:354-356.
- Rinne, J.N., and W.L. Minckley. 1991. Native fishes of arid lands: A dwindling resource of the desert southwest. USDA Forest Service General Technical Report RM-206:1-45.
- Smith, C.F., P.D. Rigas, L.K. Ham, N.R. Duet, and D.W. Anning. 1993. Water Resources Data, Arizona, Water Year 1993. U.S. Geological Survey Water-Data Report AZ-93-1, Phoenix.
- Sullivan, M.E., and M.E. Richardson. 1992. Verde River advanced identification. U.S. Fish and Wildlife Service, Arizona Ecological Services State Office, Phoenix.
- U.S. Geological Survey. 1992. Basin characteristics and streamflow statistics in Arizona as of 1989. U.S. Geological Survey, Water-Resources Investigations Report 91-4041, Phoenix.